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PRODUCTION AND MANAGEMENT OF LEACHATE FROM MUNICIPAL LANDFILLS: SUMMARY AND ASSESSMENT

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16. ABSTRACT

An assessment was made to evaluate production and management of leachate from municipal landfills for purposes of identifying practical information and techniques which may be useful to design engineers and site operators. Also assessed were: advantages, limitations, and comparative costs of various approaches for the estimation and mitigation of environmental and public health impacts, management options, and additional research needs on the generation, control, and monitoring of landfill leachates. Numerous mathematical models have been proposed for estimating leachate generation and are usually based on the water balance method. Several models have been proposed which are fairly successful in simulating the change in leachate strength with increasing landfill age or cummulative leachate volume A zone of saturation monitoring program is established to give a prompt indication of groundwater contamination.

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FOREWORD

The U.S. Environmental Protection Agency was created because of increasing public and government concern about the dangers of pollution to the health and welfare of the American people. Noxious air, foul water, and spoiled land are tragic testimonies to the deterioration of our natural environment. The complexity of that environment and the interplay of its components require a concentrated and integrated attack on the problem.

Research and development is that necessary first step in problem solution; it involves defining the problem, measuring its impact, and searching for solutions. The Municipal Environmental Research Laboratory develops new and improved technology and systems to prevent, treat, and manage wastewater and solid and hazardous waste pollutant discharges from municipal and community sources, to preserve and treat public drinking water supplies, and to minimize the adverse economic, social, health, and aesthetic effects of pollution. This publication is one of the products of that research and provides a most vital communications link between the researcher and the user community.

This report evaluates the production and management of leachates from municipal landfills for purposes of describing current practices, available control technology, comparative costs of alternative measures, management options, and mitigation of environmental impacts.

Francis T. Mayo, Director
Municipal Environmental Research
Laboratory

ABSTRACT

This report evaluates production and management of leachate from municipal landfills for purposes of identifying practical information and techniques which may be useful to design engineers and site operators. Also assessed are: advantages, limitations, and comparative costs of various approaches, for the estimation and mitigation of environmental and public health impacts, management options, and additional research needs on the generation, control, and monitoring of landfill leachates.

The quantity of leachate produced from a municipal landfill may vary considerably with management or operating practices. Numerous mathematical models have been proposed for estimating leachate generation and are usually based on the water balance method. More than one hundred different approaches are available for water balance calculations; however, few comparisons have been made in the past to identify which approach can achieve better results, or which approach is suitable for what types of landfill conditions. The applicability, accuracy, and sensitivity of water balance techniques are, therefore, greatly lacking.

The composition of municipal landfill leachate is highly variable from site to site, and major changes in leachate composition occur during the life of a landfill. Leachate strength is influenced by factors which include refuse composition, refuse density, rate of water infiltration, landfill depth, and landfill temperature. Several models have been proposed which are fairly successful in simulating the change in leachate strength with increasing landfill age or cumulative leachate volume. At present, however, the models have only been used in interpreting experimental data, and their usefulness as predictive tools in field-scale situations is either limited or yet to be tested.

The attenuation and migration of contaminants in the soil/water system are influenced by physical, chemical and biological mechanisms. Quantitative data relating to the mobility and attenuation of individual constituents by specific mechanisms are not available; consequently, considerable effort has been placed on the development of sophisticated conceptual-mathematical leachate migration models. However, due to the highly complex leachate/soil environments, no leachate migration model exists that can simulate all of the physical, chemical, and biological

processes occurring in a typical landfill system.

Controlling leachates involves preventing their formation, preventing their movement once formed, and controlling their composition. Surface water control measures such as contour grading and surface water diversion, surface sealing, and revegetation are used to minimize the quantity of water entering the landfill for purpose of reducing leachate generation. Plume management procedures involve actively altering the course of leachate movement by either adding or removing water from around the landfill. In general, a long term reduction of the strength and contaminant flux of leachates can be obtained by high water application rates, small refuse particle size, and shallow landfill depths.

The successful treatment of high strength leachates will probably require different combinations of unit treatment techniques, the appropriate combinations dictated by the character of the leachate. For a high strength leachate containing high concentrations of both organic and inorganic contaminants, a combination of biological and physical/chemical processes will be needed. Regardless of the system selected, leachate recirculation can be beneficial in reducing the organic strength and volume of leachate, as well as reducing treatment costs.

A zone of saturation monitoring program is established to give a prompt indication of ground-water contamination. The size of the landfill, hydrogeologic environment, and budgetary constraints are factors which will dictate the actual number of wells. Non-sampling methods provide for the determination of water content and water movement in the vadose zone. Incorporation of vadose zone devices can provide an early warning of potential ground-water pollution. Additionally, an effective vadose zone monitoring network could reduce or largely preclude, the requirements for ground-water monitoring.

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CONTENTS

Foreword..iii
Abstract.. iv
Figures... ix
Tables..xiv
Acknowledgement...xix

1. Introduction.. 1
2. Conclusions and Recommendations.................... 3
3. Leachate Generation................................ 7
    Introduction.................................... 7
    Factors Affecting Leachate Generation........ 7
        Precipitation........................... 10
        Surface Run-on and Runoff............... 14
        Ground-Water Intrusion and Irrigation... 14
        Refuse Decomposition.................... 16
        Codisposal of Municipal and Industrial Sludges............................. 17
        Evapotranspiration...................... 21
        Infiltration............................ 25
        Moisture Retention...................... 27
        Percolation............................. 27
    Concepts and Techniques Describing Leachate Generation.................................. 31
        Determining Precipitation............... 35
        Determining Surface Runoff.............. 36
        Determining Infiltration................ 51
        Determining Evapotranspiration.......... 56
        Determining Water Storage Capacity...... 66
        Determining Time of First Appearance of Leachate.......................... 69
        Determining Ground-Water Intrustion..... 73
    Examples of Leachate Generation Models....... 76
        Quantification of Leachate Generation Factors............................. 76
        Water Balance Calculation............... 80
        HSSWDS Model............................ 91
    Discussion.................................... 95
        Sensitivity Analysis.................... 95
        Applicability and Accuracy.............. 98
4. Leachate Composition...............................108
    Introduction....................................108
    Chemical Composition............................108

Microbiological Composition....................123
Bacteria....................................123
Viruses.....................................125
Fungi.......................................127
Parasites...................................127
Summary.....................................128
Principal Mechanisms Leading to Transfer of Refuse Mass To The Percolating Water.....128
pH..130
Redox Potential.............................130
Adsorption and Complexation.................130
Temperature.................................131
Biological Mechanisms.......................131
Summary.....................................132
Factors Affecting Leachate Composition........133
Refuse Composition..........................133
Refuse Processing...........................136
Landfill Age................................144
Rate of Water Application...................152
Depth of Leached Bed........................161
Landfill Temperature........................162
Summary.....................................165
Effects of Adding Sludges to Municipal Landfills.....................................165
Addition of WWTP Sludges....................166
Addition of Industrial Sludges..............167
Leachate Composition Models...................169
Curve Fitting Approaches....................170
Process Modeling Approaches.................173
Summary.....................................182
5. Leachate Migration...........................188
Introduction..................................188
Leachate/Environmental Interactions...........189
Soil Properties.............................189
Migration Mechanisms........................190
Migration Trends of Contaminants............200
Types of Leachate Migration Models............214
Descriptive Models..........................214
Physical Models.............................214
Analog Models...............................216
Mathematical Models.........................216
Model Distinctions............................216
Empirical or Conceptual Models..............222
Deterministic or Stochastic Models..........222
Static or Dynamic Models....................222
Spatial Dimensionality of the Model.........223
Conceptual-Mathematical Models................223
Saturated-Unsaturated Transport Models......224
Saturated-Only Transport Models.............224
Unsaturated-Only Transport Models...........224
Analtyical Models...........................224
Mathematical Solutions For Conceptual Models..229

- Analytical Methods........................ 229
- Numerical Methods......................... 229
- Summary................................... 231
- 6. Available Control Technology............ 233
  - Introduction.............................. 233
  - Leachate Volume Control................... 234
    - Ground-Water Control Measures............ 234
    - Surface Water Control.................... 253
  - Leachate Composition Control.............. 259
    - Landfill Construction and Operation Features.......................... 261
    - Leachate Recirculation................... 265
    - Addition of Municipal and Industrial Sludges.................... 267
    - Addition of Selected Sorbents............ 269
  - Leachate Collection Systems............... 270
    - Shallow Drains........................... 270
    - Well Systems............................. 271
  - Leachate Treatment........................ 272
    - Basic Leachate Treatment Methods......... 272
    - Leachate Treatment Systems............... 276
    - Treatment Costs.......................... 283
  - Summary................................... 288
- 7. Environmental Monitoring................ 290
  - Introduction.............................. 290
  - Vadose Zone............................... 290
    - Monitoring Approaches.................... 292
    - Available Monitoring Equipment and Costs............................ 292
  - Zone of Saturation........................ 311
    - Monitoring Approaches.................... 311
    - Available Monitoring Equipment and Costs............................ 313
  - Monitoring Approaches and Considerations.. 328
    - Selection of Sampling Locations.......... 328
    - Monitoring Frequency..................... 335
    - Monitoring Parameters.................... 335
    - Sample Collection Methods, Containers and Sample Preservation, and Analytical Procedures............... 337
    - Analytical Procedures and Costs.......... 342
  - Summary................................... 342
- References.................................... 344
- Appendices.................................... 383
  - A. Thornthwaite Tables........................ 383
  - B. Case Study Sites Used for Evaluation of Leachate Generation Models................................ 418
  - C. Municipal Landfill Leachate Composition.............. 420
  - D. Additional Research Needs.............................. 433

FIGURES

Number | Page
1 Factors affecting leachate volume generation.......... 9

2 Mean annual rainfall in the United States based on 40 years of record (1899-1938).................. 12

3 Mean annual snowfall in the United States............ 13

4 Mean annual effective precipitation (calculated from Figures 2 and 3).................. 15

5 Maximum non-leaching sludge to refuse ratio........... 18

6 Leachate production resulting from sludge compaction.. 19

7 Leachate production resulting from sludge compaction and decomposition.............................. 20

8 Mean annual pan evaporation............................ 22

9 Mean annual net effective precipitation (prepared from Figures 4 and 8).............................. 23

10 Relationship of rainfall, runoff, evapotranspiration, and infiltration during a rainstorm event........... 26

11 Potential infiltration for landfills in various geographical areas of the United States............. 28

12 Moisture content as a function of time for a draining soil or refuse............................ 29

13 Example showing the effect of moisture content on permeability for a clay soil....................... 32

14 Municipal landfill water balance...................... 34

15 Estimation of direct runoff amounts from storm rainfall.. 46

16 Chart for estimating 50-year-frequency peak rates of flow... 48

Number Page

17 Distribution of rainfall factors used with the modified Cook's method.............................. 49

18 Temperature versus saturated vapor pressure.......... 62

19 Temperature versus A for use in the Penman equation (Equation (28))........................... 63

20 Water storage capacity of USDA soils................ 67

21 Refuse field capacity versus refuse dry density..... 68

22 Time needed for one inch infiltration per month to bring soil from wilting to field capacity...... 72

23 Time needed for percolate to pass through the refuse.. 74

24 Estimating rate of landfill underflow............... 75

25 Flow chart of water balance calculation............. 77

26 Generalized flowchart for the hydrologic simulation model HSSWDS........................... 93

27 Leachate constituent concentration ranges........... 110

28 Relationship between refuse permeability and refuse density.................................... 138

29 Mass removal of COD versus cumulative leachate volume.. 145

30 Relationship between landfill age and leachate composition....................................... 146

31 Volatile acids found in leachates during early stages of stabilization........................... 154

32 Trends in the identified fractions of leachate TOC vs. landfill age............................. 155

33 Moisture addition and leachate response............. 158

34 Leachate total solids concentration from experimental landfills subject to different moisture application rates................................ 160

35 Leachate concentration history curves for simulated landfills of different depths.................... 163

Number		Page
36	Simulated and experimentally derived concentration histories for COD and chloride from Wigh (1979)	172
37	Single well-mixed reactor for model of leachate inorganics	176
38	Measured and simulated results of leachate total solids concentration history from Straub (1979)	178
39	Single well-mixed reactor for model of leachate organics	179
40	Measured and simulated results of leachate COD concentration history from single reactor model from Straub and Lynch (1982)	180
41	Vertically cascaded reactor model of landfill	181
42	Measured and simulated results of leachate COD concentration history for vertically cascaded reactor model from Straub and Lynch (1982)	183
43	USDA soil textural classification	191
44	Solubilities of major phosphate solids	208
45	Total soluble phosphate levels in leachates as affected by pH and alkalinity	209
46	Design specification for surface and interceptor ditches	255
47	Landfill design and operational features promoting rapid landfill stabilization	263
48	Landfill design and operational features promoting gradual landfill stabilization	264
49	COD removal efficiencies for the treatment of leachate	277
50	Percent removal of NH_3, Fe, Zn, Ni, Cd, and Cu by various treatment alternatives	278
51	Schematic of complete leachate treatment plant	280
52	Effects of added moisture on leachate COD	284

Number Page

53 Hydrogeological cycle.............................. 291

54 Generalized cost of coring in unconsolidated and consolidated formation, December, 1981........ 296

55 Pressure/vacuum lysimeter........................... 297

56 Cost of pressure/vacuum lysimeter for various depths and sampling densities, December, 1981..... 299

57 Schematic representation of tensiometer and section through the ceramic cup........................... 300

58 Cross section of tensiometer-pressure transducer assembly.. 301

59 Cost of tensiometer and electrical meter for various depths and sampling densities, December, 1981.. 303

60 Soil psychrometer................................... 304

61 Cost of psychrometer and electrical meter for various depths and sampling densities, December, 1981...................................... 305

62 Gypsum electrical resistance block.................. 306

63 Cost of multiple electrical resistance blocks and soil moisture meter for various depths and sampling densities, December, 1981................ 309

64 Equipment and principles of neutron moisture loggings.. 310

65 Cost of neutron logging for various depths and sample space densities, December, 1981............ 312

66 Determination of approximate ground-water flow direction.. 314

67 Costs of well drilling in unconsolidated formations, December, 1981........................ 317

68 Costs of well drilling consolidated formations, December, 1981...................................... 318

69 Costs of PVC casing, December, 1981................ 320

Number Page

70 Total costs of 4-, 5-, and 6-inch diameter wells in consolidated formations, December, 1981......... 322

71 Total costs of 4-, 5-, and 6-inch diameter wells in unconsolidated formations, December, 1981....... 323

72 Single screened ground-water monitoring well......... 325

73 Costs of 2-, 3-, 4-, 5-, and 6-inch diameter monitoring wells, December, 1981.................. 326

74 Well clusters.................................. 327

75 Single (A) and multiple (B) installation configurations for air-lift samplers............... 329

76 Costs of 1¼-, 1½-, and 2-inch diameter air lift samplers................................ 330

77 Hypothetical hydrologic conditions and possible location of vadose monitoring devices.............. 332

78 Cross section of Genasco Landfill and underlying materials showing a portion of the monitoring well-lysimeter network............................ 334

TABLES

Number / Page

1 Approximate Seasonal Transpiration of Selected Vegetation.......... 24

2 Generalized Water Balance Equation at a Municipal Landfill Site.......... 33

3 Runoff Coefficients for Storms of 5- to 10-Year Frequencies.......... 38

4 Runoff Coefficients for Drainage Areas with Different Topography, Soil, and Cover Conditions..... 39

5 Runoff Coefficients as Affected by Cover Material and Slope.......... 40

6 Runoff Coefficients Used by Salvato, et al., for Leachate Estimation.......... 40

7 Rainfall Limits for Estimating Antecedent Moisture Conditions.......... 41

8 Hydrologic Soil Groups Used by Soil Conservation Service.......... 42

9 Runoff Curve Numbers for Hydrologic Soil-Cover Complexes (for AMC II).......... 43

10 Runoff Curve Number (CN) Conversions and Constants.......... 45

11 Incremental W Values for Use in Cook's Method.......... 47

12 Frequency Factors for Use With Cook's Method.......... 50

13 Typical f_1 Values for Bare Soils.......... 52

14 Vegetation Cover Factor for Estimating Infiltration Capacity.......... 53

15 Values of the Constant K for Degree-Day Equation....... 54

Number Page

16 Evapotranspiration Equations.......................... 58

17 Seasonal Consumptive-Use Coefficients k in Blaney-Griddle and Blaney-Morin Equations, for Irrigated Crops in Western United States............ 59

18 Daytime-Hour Percentages, or 100p, in Blaney-Criddle and Blaney-Morin Equations......................... 60

19 Mean Possible Duration Expressed in Units of 30 Days of 12 Hr Each, or the Adjusting Factor for Potential Evapotranspiration Computed by the Thornthwaite Equation........................... 61

20 Midmonthly Intensity of Solar Radiation on a Horizontal Surface, in Millimeters of Water Evaporated per Day, or the Value R in the Penman Equation.................................... 64

21 Values of B in the Penman Equation..................... 65

22 Water Absorption Ranges for Solid Waste Components..... 70

23 Predicted Range of Absorptive Capacity of Municipal Refuse as Received at Oceanside Landfill............ 71

24 Potential Evapotranspiration Computed by Thornthwaite Method.. 81

25 Moisture Routing Through a Sanitary Landfill Soil Cover... 83

26 Moisture Routing Through Underlying Compacted Refuse (First Year).............................. 85

27 Mositure Routing Through a Landfill Soil Cover After One Year of Emplacement...................... 87

28 Moisture Routing Through Underlying Compacted Refuse (Second Year)............................. 88

29 Moisture Routing Through Underlying Compacted Refuse (Third Year).............................. 89

30 Water Balance Calculation for a Landfill in Cincinnati, Ohio................................ 90

31 Water Balance Calculation for a Landfill in Los Angeles, California.......................... 92

Number Page

32 Leachate Production at Blue Valley Sanitary Landfill Under Different Assumptions, in Inches of Water.. 96

33 Summary of Sensitivity Study Results.................. 97

34 Comparisons Among Leachate Estimation Methods......... 100

35 Methods Used for the Evaluation of Leachate Generation.................................... 106

36 Comparisons of Leachate Generation Between Measured and Calculated Results..................... 107

37 Organic Compounds or Classes Identified in Landfill Leachates.. 122

38 Bacteria and Bacteriologic Indicators Identified in Municipal Leachates............................ 124

39 Cumulative Mass Removal of Refuse Pollutants by Laboratory Columns and Mass Removal by Extraction Testing................................ 135

40 Cumulative Masses of Leachate Parameters per kg Dry Refuse on a Basis of Equal Leachate Volume.......... 140

41 Sustained Maximum Concentrations of Leachate Contaminants (from Streng, 1976).................... 168

42 Total Available Mass Removal (from Wigh, 1979)........ 173

43 Summary of Leachate Composition Models................ 184

44 Soil Organisms.. 192

45 Important Solubility Products (K_{sp}) of Trace Metals... 195

46 Theoretical Redox Conditions for Redox Couples........ 202

47 Major Phosphate Controlling Solids.................... 207

48 Factors that Influence the Movement of Viruses in Soils.. 213

49 Example Models and Their Classification into Different Groups.................................. 215

Number Page

50 Partial List of Available Transport Models for Application to Ground-Water Quality Problems........ 217

51 Explanation of Symbols used in the Mass Transport and Flow Equations.................................. 227

52 Partial List of Equations Used to Describe Adsorption Reactions.............................. 228

53 Properties of Commonly Used Liners.................... 236

54 Typical Characteristics of Synthetic Liners........... 237

55 Selection Criteria for Synthetic Liners............... 239

56 Synthetic Layers for Asphalt, Clay, and Bentonite..... 242

57 Costs of Flexible Polymeric Membrane, Plastic, and Rubber Liners................................ 243

58 Cost Estimations of Soil, Admix Materials, and Asphalt Membrane Liners......................... 244

59 Costs of Ground-Water Control Measures................ 245

60 Relative Costs of Grout............................... 249

61 Leachate Plume Control................................ 251

62 Surface Water Control................................. 256

63 Costs of Surface Seals................................ 258

64 Grasses and Legumes Commonly used for Vegetation...... 260

65 Leachate Recirculation and Collection Technologies.... 266

66 Kinetic Parameters for Aerobic Treatment of Leachate (High Strength) and Domestic Wastewater............ 274

67 Summary of System 1 Operation Data.................... 279

68 A Summary of Cost Estimates for Leachate Treatment.... 286

69 Estimated Comparative Annual Leachate Treatment Costs for Recirculated and Equivalent Non-Recirculated Landfills.............................. 287

Number Page

70 Comparative Savings of Recirculated Case over Non-Recirculated Case.............................. 288

71 Basic Parts and Costs of a Veihmeyer Soil Sample..... 295

72 Advantages and Disadvantages of Zone of Saturation Monitoring Devices...................... 321

73 Recommendations for Sampling and Preservation of Samples.. 339

74 Costs of Water Quality Analyses, December, 1981. 343

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SECTION 1

INTRODUCTION

In 1990, a projected 295 to 341 million metric tons of municipal refuse will be produced annually in the United States (Doggett, et al., 1980). The disposition of this huge volume of material in a sanitary landfill conveys with it an inherent potential for pollution of indigenous water resources. With the increasing demands on our land and water resources, the protection of these resources from the impacts posed by municipal solid waste disposal is imperative.

A major environmental impact, resulting from the disposal of municipal solid waste into a sanitary landfill, occurs when water passing through the refuse accumulates various contaminants. This percolate or leachate may enter underlying ground waters and seriously degrade the water quality of the aquifer.

Potential pollution from a refuse disposal area must be considered in its design and location. Understanding of the factors and conditions involved in the production of leachate is necessary for its elimination and control. A major goal in planning, designing, and operating a sanitary landfill is to minimize all sources of water reaching the refuse so that leachate production is avoided. Leachate production can be minimized or nearly eliminated by preventing water contact with the refuse by the use of surface and subsurface drainage and properly selected cover material that is graded and seeded with a high evapotranspiration crop.

In some cases it will not be economically feasible to eliminate leachate production from these refuse areas and treatment of the leachate will be necessary. The processes to treat this material effectively have not been studied in detail, but either chemical or biological treatment should be effective. It is also possible that a combined treatment system in which the major treatment is done by chemical and physical means, with the polishing being done in lagoons or other biological processes, would be the most economical. In all cases, however, minimizing the quantity of leachate production, as well as protection of the ground and surface waters, should be the major goal.

An extensive body of literature is available that describes sanitary landfill leachate production, composition, migration, control, and monitoring. Out of these works have evolved design and operational criteria which are primarily empirical in nature and which may or may not have a relationship to environmental conditions. Various studies concerning sanitary landfill behavior have been conducted in recent years to better understand them and to delineate and define significant management options. However, a review of existing literature has revealed a number of areas requiring additional research for technological development.

The objectives of this study are: (1) to clarify the understanding of leachate production and management options through a critical review and analysis of existing information, (2) to identify practical information and techniques which can be used by design engineers and site operators, (3) to delineate weaknesses in the available data base, and (4) to identify techniques which may be useful in the estimation and mitigation of environmental and public health impacts caused by leachate generation. Specific objectives are to provide documentation of current methodologies, advantages and limitations of various approaches, comparative costs, and additional research needs on the generation, significances, and associated costs of controlling and monitoring leachate from municipal landfills. This information is presented in terms of the following subject areas:

- Leachate generation;
- Leachate composition;
- Leachate migration;
- Available control technology; and
- Environmental monitoring.

The presentation of each of these subjects is based upon a review of the available literature and site histories.

SECTION 2

CONCLUSIONS AND RECOMMENDATIONS

CONCLUSIONS

A critical evaluation of production and management of leachate from municipal landfills was performed to identify practical information and techniques that might be useful for landfill design and operational purposes. This information, including economic evaluation, is important in efforts to estimate, reduce, and control the risks of environmental and public health impacts caused by leachate production. Existing literature revealed a significant volume of information on the generation, composition, significance and costs of controlling leachates, as well as an absence of data necessary to the development of leachate predictive techniques. Report conclusions are presented in the order of their discussion in the text.

The quantity of leachate produced from a municipal landfill may vary considerably with management or operating practices, depending on whether leachate is viewed as a short or long-term problem. Operational factors which affect leachate quantity may include: cover material handling, watering prior to compaction, watering following compaction, daily variation in cell construction, and variation in waste composition (e.g., municipal refuse alone or codisposal of municipal refuse and sewage sludge or industrial wastes; milled or unmilled refuse).

Numerous mathematical methods have been proposed for a quantitative estimation of the volume of leachate generated from landfills and are usually based on a mass balance approach (i.e., water balance method). Components of these models are relatively easy to obtain but other model variables such as the surface runoff coefficients, runoff curve number, and evapotranspiration from the landfill surface are more difficult to develop. Limited field data exists for verification for many of the leachate generation models. Therefore, the applicability, accuracy, and sensitivity of leachate generation models, is largely unknown.

The composition of leachate produced from a municipal landfill is also highly variable, depending on factors such as refuse composition, refuse processing, landfill age, the rate of infiltration, landfill depth, and landfill temperature. The quality of leachate can be controlled to a large degree. Shredding and baling of refuse, landfill depth, and the rate of water application to the landfill surface can influence the rate at which contaminants are released from refuse, and can determine whether leachates have a long or short-term pollution potential. Based on empirical data garnered from a number of studies addressing landfill and leaching behavior, several models have been developed which describe leachate quality as a function of time or cumulative leachate generation. The most promising of these models attempts to simulate the physical/chemical and biological processes which occur during leaching. Presently, however, the leachate composition models are useful only in the interpretation of experimental results, rather than finding application to field-scale problems.

Leachate migration models are based upon constituent mass transport and water flow equations. Numerous models are available for predicting chemical and physical migration; biological models, however, are generally lacking. While a large number of conceptual-mathematical models exist, none are universally applicable for the simulation of all of the physical, chemical, and biological processes that are operative in a typical waste disposal system. The complexity of these processes, which operate in a simultaneous and interactive manner, are probably such, that the development of a generic model would be impractical because the resulting program would undoubtedly become so large and complex that the cost of operating it would be exorbitant.

Leachate control technology is a relatively well-developed methodology for the management of landfill leachates. Various ground-water and surface water control approaches are available for the control of leachate into or out of the fill. Approaches in which dewatering or counter pumping (injection) is practiced appear to be effective control measures, although the long term energy costs for pumping are restrictive. Accelerated stabilization processes such as leachate recycling and nutrient addition are promising, but require additional study. Extensive liner technology is available; however, field verification studies have, in general, been inconclusive regarding anticipated liner lifetime.

Factors which affect leachate composition and are amenable to control are: (1) physical characteristics of refuse, (2) rate of water application or leachate recycle, and (3) landfill depth. In general, shredded refuse, higher water/leachate application rates, and shallower landfill depths are conducive to increased refuse stabilization and, ultimately, a dilute leachate.

The successful treatment of municipal landfill leachates has been demonstrated. The selection of an appropriate treatment depends on the character of the leachate and the proximity to a wastewater treatment facility. Whereas conventional wastewater treatment poses a legitimate treatment option for landfills close to a treatment plant, leachates of landfills distant to a plant may be treated by aerated lagoon, land application, or by conventional biological and/or physical/chemical treatment techniques. Regardless of the system selected, leachate recirculation can be beneficial in reducing the organic strength and volume of leachate, as well as reducing treatment costs.

Vadose zone monitoring has received little attention in contrast to the advanced technology for monitoring in the zone of saturation. This discrepancy reflects, to a large extent, the greater complexity of flow in the vadose zone, compared to saturated flow, and the related problem of obtaining a representative sample for analyses. Incorporaton of vadose zone devices can provide an early warning of potential ground-water pollution. If remedial measures are implemented prior to the onset of extensive ground-water contamination, the associated renovation costs could be reduced significantly. Additionaly, an effective vadose zone monitoring network could reduce or largely preclude, the requirements for ground-water monitoring. The savings in costs for construction of ground-water wells could be significant, particularly, in western regions where water tables are often hundreds of feet deep.

RECOMMENDATIONS

Review of the literature and case study materials revealed a number of areas that warrant further study. These are summarized in the following discussion.

Leachate generation cannot be accurately quantified due to the inadequacy of much of the data concerning the availability of water, landfill surface conditions, and refuse conditions. Therefore, additional research should be conducted regarding such variables as water contribution from refuse degradation, refuse permeability, evapotranspiration, surface runoff coefficients, and runoff curve numbers. The acquisition of reliable data may be applied to existing models for improved predictive capabilities or may be used to derive a modified technique for prediction of leachate generation.

With regard to leachate composition, the persistence and viability of enteric bacteria, viruses, and parasites in landfills and leachates require further study. Because the leaching behavior of heavy metal contaminants is highly variable, additional study is required on the roles of complexaton, adsorption, and precipitation in influencing heavy metal mobility. Also, leachate composition modeling efforts clearly need a broader empirical base in the areas of solid waste hydraulic properties, contaminant leaching patterns, and microbial activity.

Additional research should be directed towards the development of simulation technology for leachate migration models. A better understanding of the different mechanisms operative in waste disposal systems is required in order to construct more efficient and reliable simulation models. In particular, modeling efforts should be concerned with single-ion, key-parameter models, rather than complex approaches which are both unwieldly and impractical. Further research should be directed towards the development of biodegradation models which would be of great value in predicting the stabilization rate of landfill refuse. Attention should also be given to the development of a simplified chemical/physical model which incorporates the effects of soil chemical and physical properties.

Leachate control techniques which require additional study include the effects of nutrient and alkaline industrial waste injection into a landfill for the purpose of accelerating the stabilization process. Additional work should also include the effectiveness of natural and synthetic sorbent materials in removing contaminants from leachate. Leachate recycling should also receive attention since it appears to offer both increased stabilization and an effective leachate treatment scheme. The effectiveness of a leachate withdrawl and injection well system should also be investigated.

Vadose zone monitoring requires additional study to better define both the dominant phenomena occurring in this zone as well as optimum monitoring equipment. The integration of passive or non-sampling vadose zone devices should be studied to determine their possible use. Sample retrieval devices for both the vadose and zone of saturation should be examined to determine their effects on obtaining a representative sample. Attention should also be given to potential environmental impacts of surface run-off from sanitary landfills.

SECTION 3

LEACHATE GENERATION

INTRODUCTION

A quantitative description of leachate volume generation from sanitary landfills is essential for determining effects on water quality and the value of control methods. Methods of estimating leachate generation have received considerable attention. Field tests, physical and mathematical models, and monitoring of actual installations have all been performed with the intention of increasing the accuracy of these predictive methods. Because of the numerous factors involved in leachate generation; however, estimates are only accurate within an order of magnitude (Duvel, et al., 1979). This lack of accuracy is due primarily to the inability to measure the surface runoff coefficients and evapotranspiration of the landfill site and adjacent area. Other problems include estimation of the in situ permeabilities (i.e., hydraulic conductivities) of wastes, and of underlying and cover soils. All of these factors contribute to the complex problem of estimating leachate volume.

In this section, the various factors affecting leachate generation will be discussed and pertinent generation concepts and measurement techniques will be evaluated. Examples of selected leachate generation models are provided. Calculations are compared to the actual field data where this is feasible. The advantages and limitations of existing models are discussed.

FACTORS AFFECTING LEACHATE GENERATION

Leachate volume generation at a landfill site is dependent on many factors; in general, it is determined by the following four conditions (Merz, 1954; Remson, et al., 1968; Qasim, et al., 1970; Fungaroli, 1971; Rover, et al., 1973; Caffrey, et al., 1974; Fenn, et al., 1975; Dass, et al., 1979; Fungaroli et al., 1979; and Lutton, et al., 1979):

- Availability of water;
- Landfill surface conditions;
- Refuse conditions; and

- Underlying soil conditions.

The major factors which affect the above conditions are summarized in Figure 1.

Factors affecting the availability of water include direct precipitation, surface run-on, ground-water intrusion, irrigation, refuse decomposition, and codisposal of liquid waste or sludge with refuse. Of these water sources, the primary contributor is direct precipitation (Caffrey, et al., 1974; Fenn, et al., 1975; Dass, et al., 1977). Precipitation in adjacent areas can be channeled onto the landfill site (i.e., surface run-on). If refuse is placed in or just above the ground-water table, a ground-water mound may saturate the refuse and result in considerable quantities of leachate. Irrigation applied to the landfill surface may also contribute to leachate generation. This is especially true if the completed landfill is used as a golf course or for similar recreational purposes (Fenn, et al., 1977). Water from refuse decomposition is another contributor (Remson, et al., 1968; Rovers, et al., 1973; Dilaj, et al., 1975). Co-disposal of liquid wastes or sludge in municipal landfills may also contribute to leachate generation (Stone, 1974; Parkhurst, 1978). All of the above mentioned leachate sources will be discussed in detail later in this section.

Water reaching the landfill surface by precipitation, surface run-on, or irrigation may either evaporate or transpire, infiltrate through the landfill surface, or leave the site as surface runoff. Pathways available to the water and the distribution of water among them will depend principally upon the landfill surface conditions. Surface conditions which may affect leachate generation include vegetation, cover material (type, dimension, compaction, permeability, moisture content, etc.), surface topography, temperature, humidity, and wind speed above the landfill (Chow, 1964; Remson, et al., 1968; Rovers, et al., 1973; Caffrey, et al., 1974; Dass, et al., 1977; and Lutton, et al., 1979).

Once field capacity[*] of the surface cover material is attained, leachate will percolate through the refuse. Retention and transmission characteristics of the refuse control the percolation rate. Leachate may be channelled through the refuse and dispersed by refuse intermediate cover layers, or may seep through the pores of the refuse. If channelling does not occur, leachate will not be produced in a landfill until at least a portion of the refuse reaches field capacity. Any additional moisture will then cause leachate movement. Underlying soil

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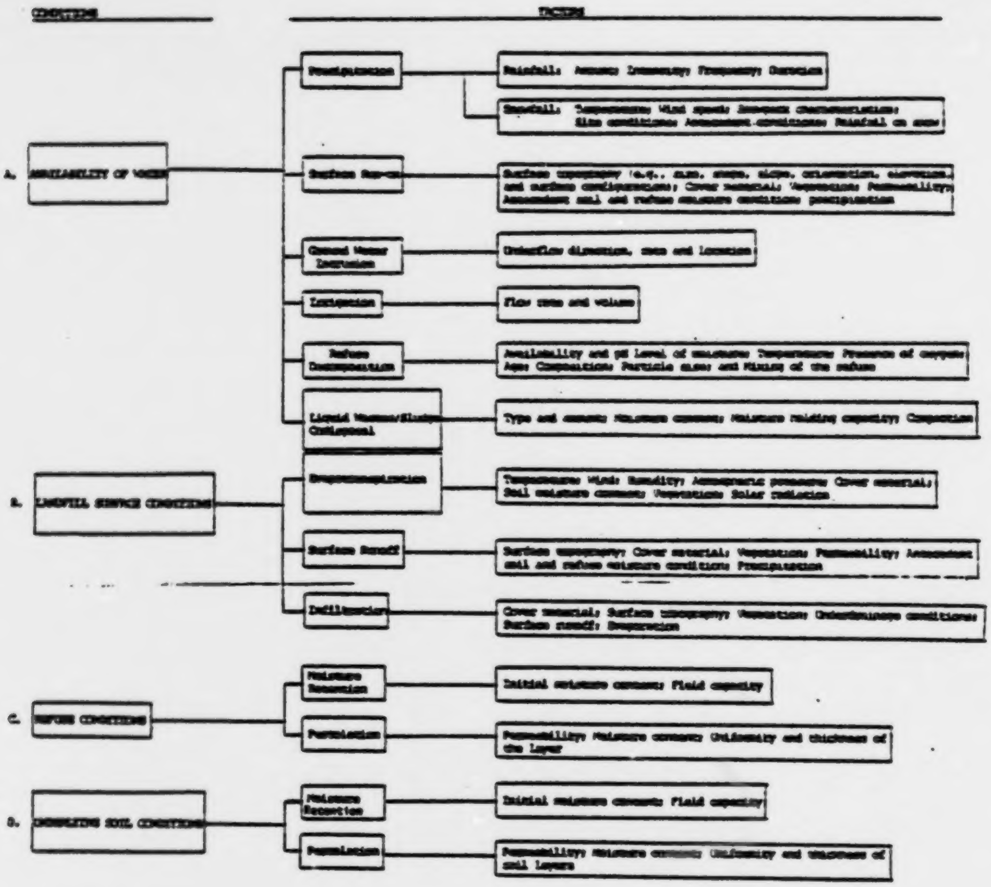
[*] Field capacity is defined as the maximum moisture content which a soil or a solid material can retain in a gravitational field without producing continuous downward percolation.



CONDITIONS
FACTORS
A. AVAILABILITY OF WATER
Precipitation
Rainfall: Amount; Intensity; Frequency; Duration
Snowfall: Temperature; Wind speed; Snowpack characteristics; Site conditions; Antecedent conditions; Rainfall on snow
Surface Run-on
Surface topography (e.g., size, shape, slope, orientation, elevation, and surface configuration); Cover material; Vegetation; Permeability; Antecedent soil and refuse moisture conditions; precipitation
Ground Water Intrusion
Underflow direction, rate and location
Irrigation
Flow rate and volume
Refuse Decomposition
Availability and pH Level of moisture; Temperature; Presence of oxygen; Age; Composition; Particle size; and Mixing of the refuse
Liquid Wastes/Sludge Codisposal
Type and amount; Moisture content; Moisture holding capacity; Compaction
B. LANDFILL SURFACE CONDITIONS
Evapotranspiration
Temperature; Wind; Humidity; Atmospheric pressure; Cover material; Soil moisture content; Vegetation; Solar radiation
Surface Runoff
Surface topography; Cover material; Vegetation; Permeability; Antecedent soil and refuse moisture conditions; Precipitation
Infiltration
Cover material; Surface topography; Vegetation; Underdrainage conditions; Surface runoff; Evaporation
C. REFUSE CONDITIONS
Moisture Retention
Initial moisture content; Field capacity
Percolation
Permeability; Moisture content; Uniformity and thickness of the layer
D. UNDERLYING SOIL CONDITIONS
Moisture Retention
Initial moisture content; Field capacity
Percolation
Permeability; Moisture content; Uniformity and thickness of soil layers

Figure 1. Factors affecting leachate volume generation.

conditions can modify both the rate and amount of leachate generation when soils underlying and surrounding the site have lower permeabilities than the cover soils and refuse. Details of the leachate percolation through the refuse and the effects of underlying soils will be discussed later in this section.

The quantity of leachate produced may also differ considerably with management or operating practices (Duvel, et al., 1979). Operational factors may include cover material handling, watering prior to compaction, daily variation in compaction and cell construction, and variation in waste composition (e.g., municipal refuse alone or municipal refuse plus sludge or industrial wastes; milled or unmilled refuse; etc.). Basically, these factors, may affect the conditions of cover material and refuse fill, and; therefore, affect the leachate generation. Specific information describing the effects of management and operating practices, on leachate generation; however, is not available.

Precipitation

In most cases, precipitation will be the principal source of leachate, provided the contribution by irrigation, groundwater intrusion, or liquid wastes codisposal, is minimal (Fenn, et al., 1977). Precipitation includes rainfall and snowfall. Rainfall can affect leachate generation more directly and rapidly than snowfall (Caffrey, et al., 1974; Fenn, et al., 1975; and Dass, et al., 1977).

Four rainfall characteristics which affect leachate generation are amount, intensity, frequency, and duration* (Chow, 1964). Rainfall is usually used to represent the total quantity of rain water reaching the ground surface during a certain time period (e.g., a month or a year) for a given location. This rainfall amount may result from a single storm or multiple storms. A number of rainfall analyses in terms of amount are generally available in the forms of mean-annual-, mean-seasonal-, mean-monthly-, and mean-weekly-rainfall data (Chow, 1964). Many gross estimations of leachate generation are on the monthly basis (e.g., Remson, et al., 1968; Fenn, et al., 1975; and Dass, et al.

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* Amount: Rainfall amount is the quantity of rain reaching the ground surface in terms of depth of water (e.g., mm. or in.).
Intensity: Rainfall intensity is the amount of rain per unit time period (e.g., mm/hr or in./hr).
Frequency: Rainfall frequency is the repeated occurrence of a certain rainfall characteristic at a given interval.
Duration: Rainfall duration is the duration of a storm (e.g., min or hr).

1977). Such estimation methods for leachate generation usually neglected the effects of rainfall intensity, frequency, or duration. Rainfall intensity was found to influence the impact of raindrops on the surface soil particles. Such an effect could change the infiltration rates (Chow, 1964) and thus change the leachate quantity produced. Rainfall frequency and duration also could affect leachate generation through their influence on infiltration and surface runoff. For example, an annual rainfall of 10 in. in the southwestern United States probably will not produce any significant leachate if such a rainfall amount occurred during numerous storms spread over the entire year period. But leachate could very easily be generated if such a rainfall amount is caused by a few deluge-type rainstorms lasting hours or days. The quantitative information on the effects of such rainfall characteristics on leachate generation; however, is greatly lacking.

Rainfall varies with both the physiographical setting and the season and is well-documented in most hydrologic literature. Regional effects, such as latitude, mountains, large lakes, and oceans, influence rainfall patterns. Dry summers on the West Coast, the stability of monthly precipitation in New England, the winter and spring maxima from the Central Gulf to the Ohio Valley states, and the sudden transition from June drought to July thunderstorms in Arizona, are notable examples (Chow, 1964; and Viessman, et al., 1977). Leachate quantities from rainfall are likely to be less in the western states than in the east (refer to Figure 2).

Significant variations in rainfall may occur in certain localized areas, especially in mountainous regions, requiring caution in using generalized rainfall data or hydrologic maps. An abnormally wet or dry year may occur which also cannot be shown by generalized maps or data. In using rainfall data for leachate generation calculations, it is advisable to seek data specific to the landfill site. Methods of determining rainfall are discussed later in this section.

Snowfall on the landfill may also yield water for subsequent leaching. Snowfall varies with season and location. Figure 3 depicts the mean annual snowfall in the U. S. In estimating the availability of snowfall as a leaching source, the amount of snowfall, the sublimation of snow, and the melting rate of snow must be considered. The determination of the portion of the snowpack lost by sublimation is very difficult. The sublimation is probably small during the active snowmelt period when dew points* are above 0°C (32°F) (Chow, 1964). In many mountainous

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* Dew point: Dew point is the temperature at which an air mass just becomes saturated if such air mass is cooled at constant pressure and without moisture addition or removal.

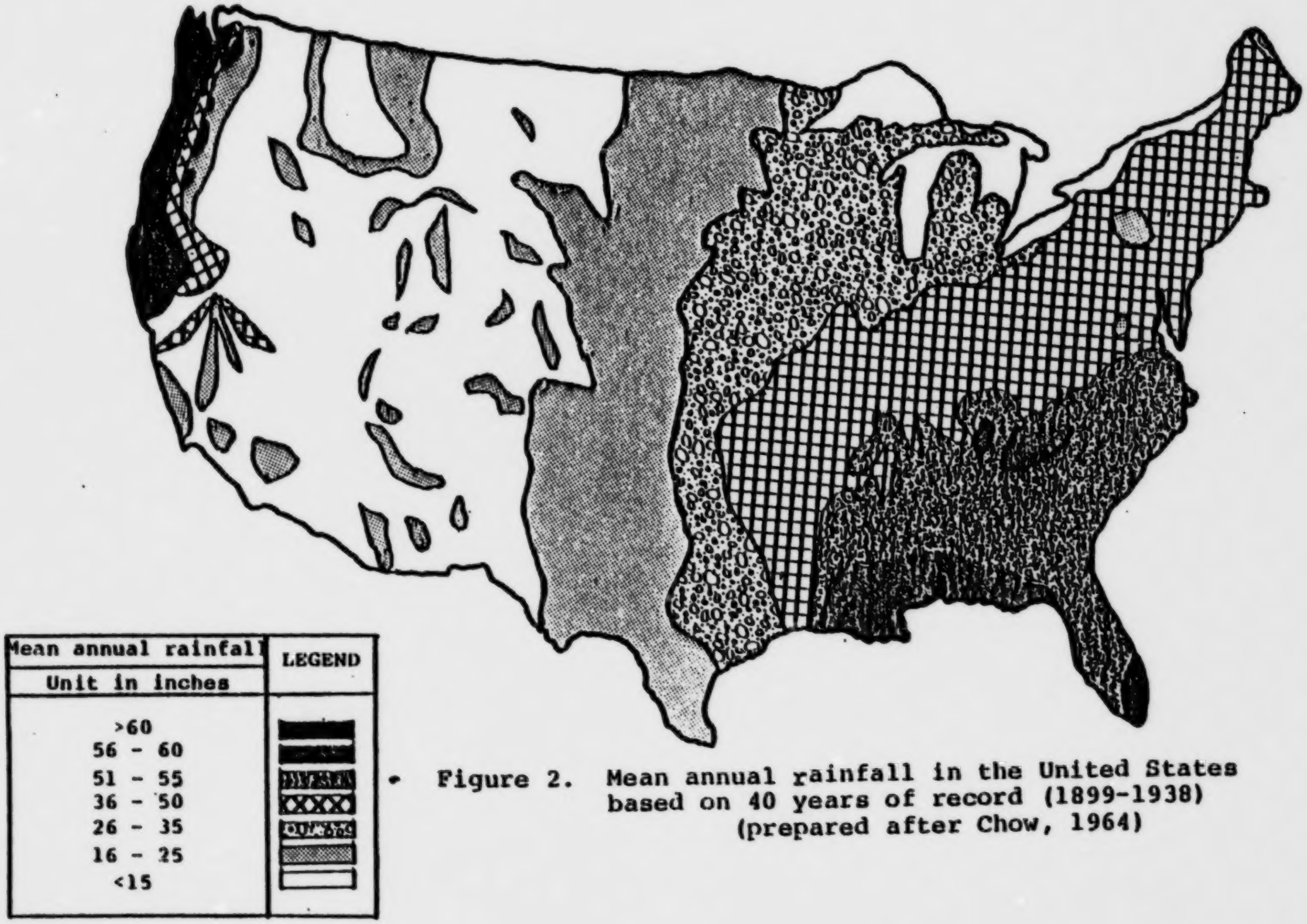
Mean annual rainfall
Unit in inches
LEGEND
>60
56 - 60
51 - 55
36 - 50
26 - 35
16 - 25
<15

Figure 2. Mean annual rainfall in the United States based on 40 years of record (1899-1938) (prepared after Chow, 1964)

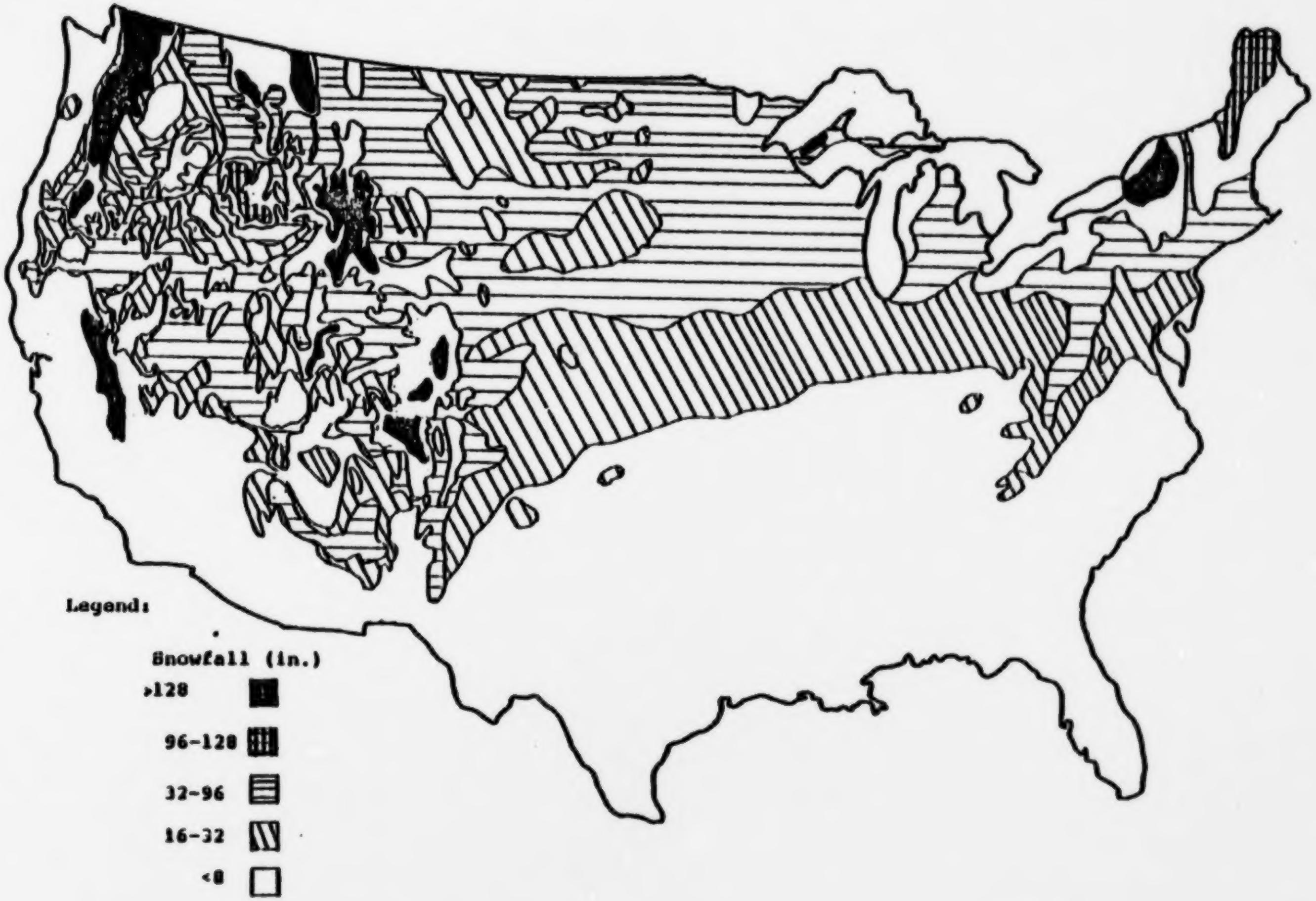
Legend:
Snowfall (in.)
>128
96-128
32-96
16-32
<8

Figure 3. Mean annual snowfall in the United States (Prepared after Chow, 1964).

areas during the winter, dew points are low and the total evaporation loss from the snow pack in areas of low snow accumulation may be significant (Chow, 1964). In the winter and early spring, before snowmelt runoff begins, sublimation losses can be assumed to be about 13 mm (0.5 in) of water equivalent per month (Chow, 1964). The melting rate of snow can be affected by temperature, wind speed, snowpack characteristics (the ripening of snow), site conditions, antecedent snowfall conditions, and rainfall. Melting usually occurs at rates considerably below the infiltration capacity of unpacked soils (Chow, 1964). Therefore, surface runoff from snowmelt may be rare. However, when the landfill surface is steep and compacted tightly which reduces infiltration, sudden changes in temperature may result in surface run-off from snowmelt. In general, it is suggested that 250 mm (10 in) of snow is equivalent to 25 mm (1 in) of rainfall (Linsley, et al., 1972). A mean annual effective precipitation* in the U. S. is calculated on this basis and is shown in Figure 4. This figure is a modification of Figures 2 and 3 as presented previously, and can be used to indicate the potential relative source of leachate in various geographic areas in the United States.

Surface Run-On or Runoff

Major land surface conditions affecting surface run-on or runoff include surface topography, cover material, vegetation, soil permeability, antecedent soil moisture, and artificial drainage (Chow, 1964; Caffrey, et al., 1974; and Dass, et al., 1977). Surface topography (size, shape, slope, orientation, elevation, and surface configuration) control flow at the surface. Among these topographic factors, slope may be most significant. Surface soil material, soil permeability, and antecedent soil moisture affect infiltration rates which in turn can affect surface run-on or runoff. Details of these effects are discussed later under "infiltration". Landfill surface vegetation also has significant effects on surface run-on or runoff. The significance of this influence depends on the species composition, age, and density of the vegetation, as well as the season. Quantitative data showing the effects of slope, cover material, and vegetation, on the surface runoff are discussed in the next subsection entitled "Concepts and Techniques Describing Leachate Generation."

Ground-water Intrusion and Irrigation

Landfill leachate from ground-water intrusion will occur if the base of the landfill is below the ground-water table. The volume of saturated refuse, contact time, and flow direction, will affect leachate production. Precise measurement of underflow is not feasible. Detection of underflow and reasonable

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* Effective precipitation = rainfall + 1/10 snowfall.

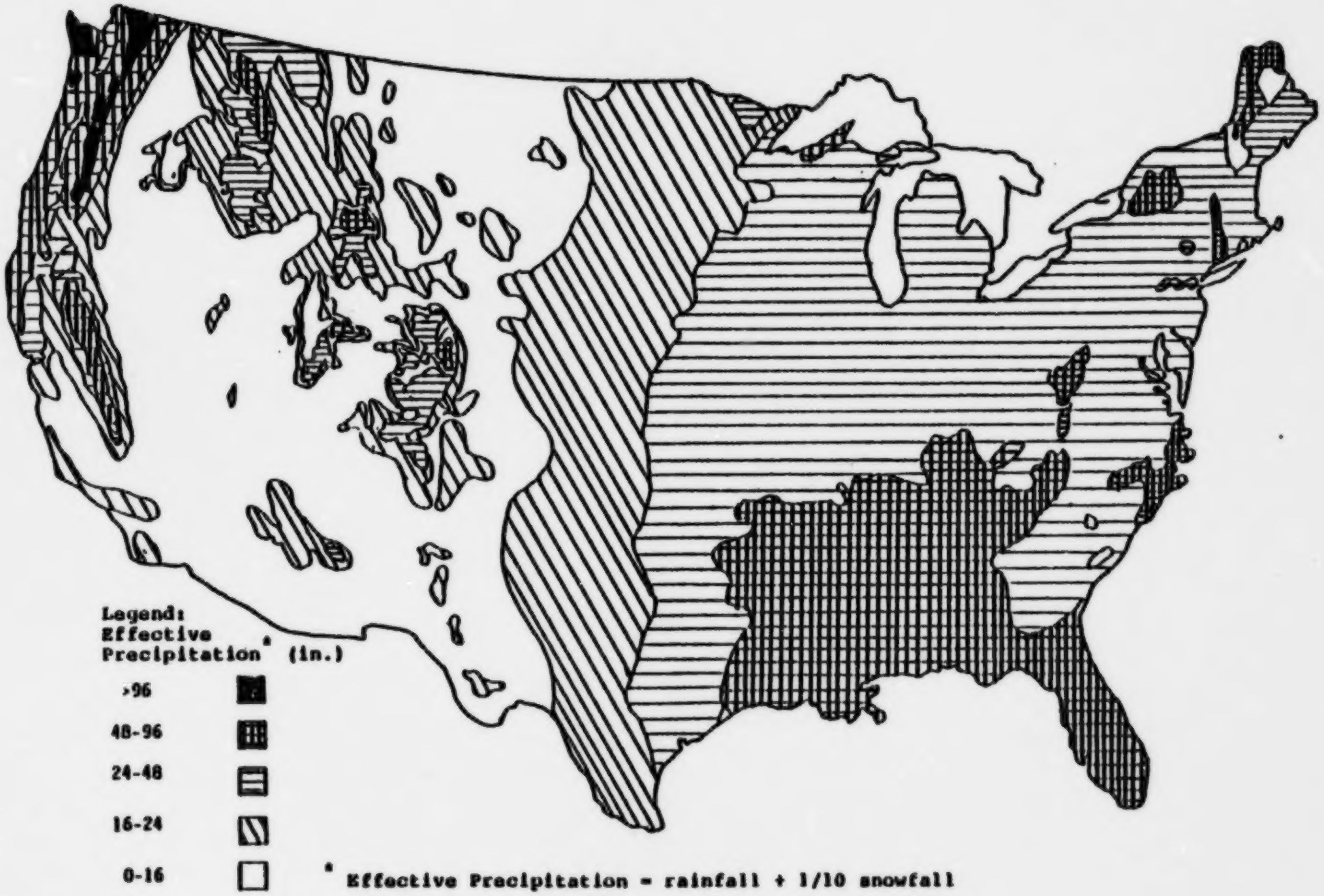
Legend:
Effective
Precipitation* (in.)
>96
48-96
24-48
16-24
0-16
* Effective Precipitation = rainfall + 1/10 snowfall

Figure 4. Mean annual effective precipitation (calculated from Figures 2 and 3).

approximation of flow rates are possible; however, with a hydrological investigation and calculations using Darcy's Law (Chow, 1964; DeWiest, 1967; and Fenn, et al., 1977). Quantification techniques for the ground-water intrusion estimation are shown later entitled "Concepts and Techniques Describing Leachate Generation".

Total amounts of irrigation water applied to a landfill can be measured with a flow meter. Because the water loss caused by surface runoff during irrigation is negligible, the leachate produced is the difference between irrigation input and evapotranspiration.

Refuse Decomposition

Microbial decomposition of the biodegradable organics in refuse can generate water, which will contribute to the refuse moisture for leachate production. Both aerobic and anaerobic reactions may generate biochemical water (California State Water Pollution Control Board, 1961; and Boluche, 1968):

$$\text{(Organic Matter)} + \text{(Oxygen)} \xrightarrow{\text{Aerobes}} \text{(Water)} + \text{(Carbon Dioxide)} + \text{(Acids)} + \text{(Inorganic Minerals) (e.g., } PO_4^{\equiv}\text{, } SO_4^{=}\text{, } NO_3^{-}\text{, or } NH_3\text{)} \quad (1)$$

$$\text{(Organic Matter)} \xrightarrow{\text{Anaerobes}} \text{(Water)} + \text{(Methane)} + \text{(Decomposed Organics)} + \text{(Carbon Dioxide)} + \text{(Ammonia)} + \text{(Inorganic Minerals)} \quad (2)$$

The extent and rate of microbial activity and consequent water generation depend mainly on the amount and pH of interstitial moisture, temperature, presence of oxygen, the composition and particle size of the refuse, the type of organisms present, and the degree of refuse mixing (Remson, et al., 1968; Rovers, et al., 1973; Caffrey, et al., 1974; and Dilaj, et al., 1975). In general, decomposition is faster under aerobic than anaerobic conditions. California State Water Pollution Control Board (CSWPCB, 1961) reported that under aerobic conditions, decomposition may reduce volatile matter and carbon concentrations by 50 percent in a year. For anaerobic conditions, although precise data is not available, the decomposition rate is known to be substantially lower. Qualitatively, the normal series of events would begin with an initial rapid aerobic decay of the refuse. The refuse would gradually become anaerobic as easily digested substances are depleted and slower heat production reduces convectional transport of oxygen. After a period of anaerobiosis, which may last several years, decay slows, and the aerobic processes may be supported by the oxygen diffusing into the fill (SCWPCB, 1961). The rate at which water is produced by decomposition would vary as these events occur.

Microbial phenomena occurring in a landfill can be described qualitatively. However, the present information is not adequate to calculate rates of refuse degradation and water generation (Remson, et al., 1968; Rovers, et al., 1973; and Dilaj, et al., 1975). Research conducted by the California State Water Pollution Control Board (1961) measured decomposition water production in a municipal landfill operated by the City of Riverside. This study found that decomposition water may reach 42 mm/m (0.5 in./ft) of refuse. No other data for biochemical water production are available. Based on the above measured decomposition water, it can be concluded that leachate generated from refuse decomposition is very minor compared to other sources of leachate.

Codisposal of Municipal and Industrial Sludges

Codisposal of municipal and industrial sludges in municipal landfills may contribute significant amounts of leaching water. The volume of water available for leaching may be affected by the type and amount of sludge, moisture content, moisture holding capacity, and the effect of compaction and decomposition on the release of water from sludge (Charlie, et al., 1979; and Parkhurst, 1978).

Experimental works performed by the Los Angeles County Sanitation District (Parkhurst, 1978) provides information on the generation of codisposal leachates as function of sludge/refuse ratio, sludge moisture content, landfill depth, and effects of compaction and decomposition (Figures 5 to 7). Figure 5 shows the maximum non-leaching sludge to refuse (wet weight) ratio as a function of sludge solids content at various landfill depths. The data demonstrates a large dependence of the maximum non-leaching sludge to refuse ratios on the depth of the disposal mixture. For example, at 25 percent solids, a sludge cake can be placed at a 0.93 to 1 ratio at the 7.5 meter depth without leachate generation. However, at the 100 meter depth, the non-leaching ratio was reduced to 1 part sludge to 9 parts refuse. This phenomenon is mainly due to the pressure effects resulting from the surcharge provided by successive lifts (Parkhurst, 1978). Figure 6 illustrates the leachate release from the codisposal landfill caused by surcharge loads when the non-leaching ratio is exceeded. Figure 7 further depicts the ultimate leachate production resulting from both surcharge loads and decomposition.

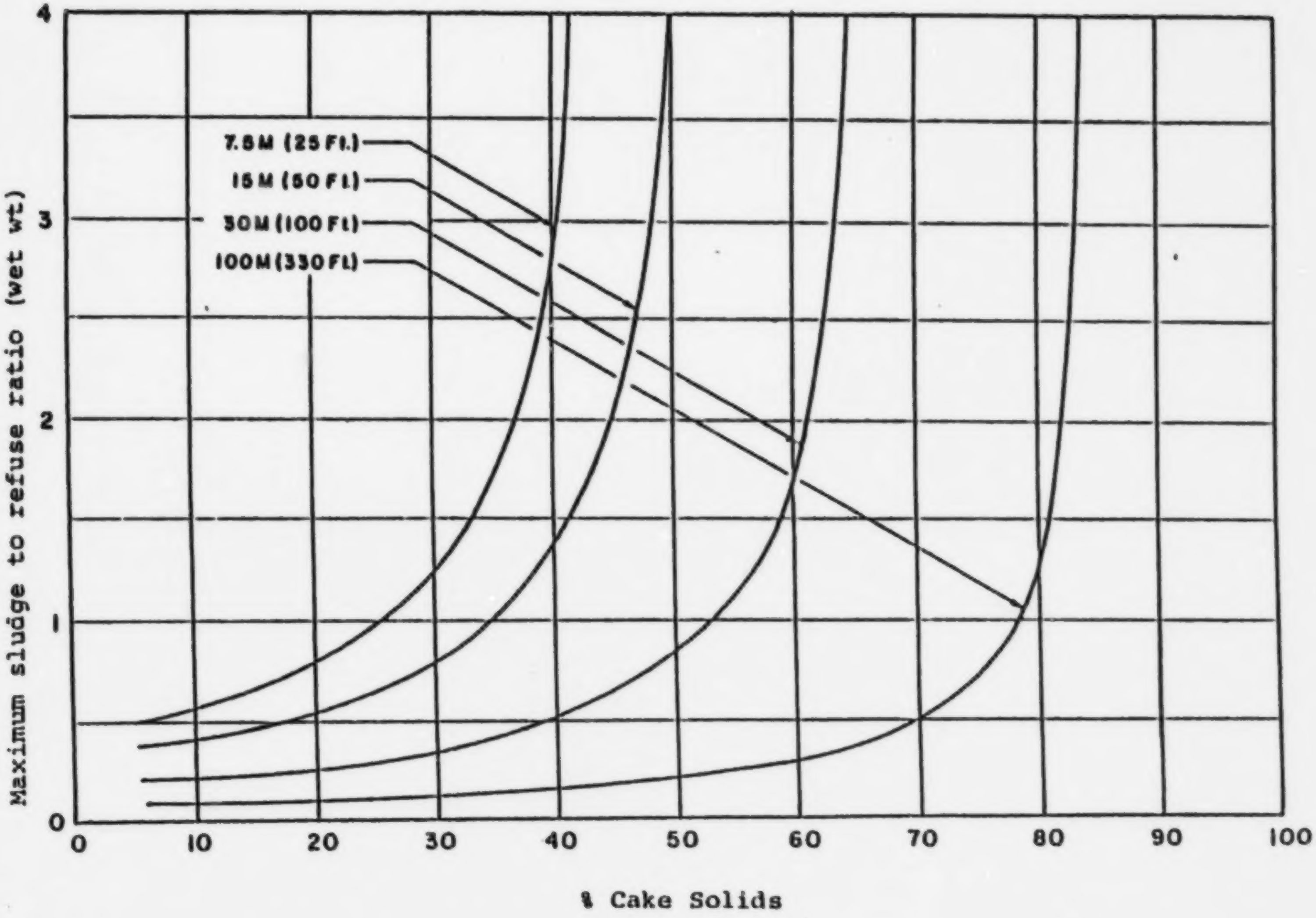

Figure 5. Maximum non-leaching sludge to refuse ratio (Parkhurst, 1978).

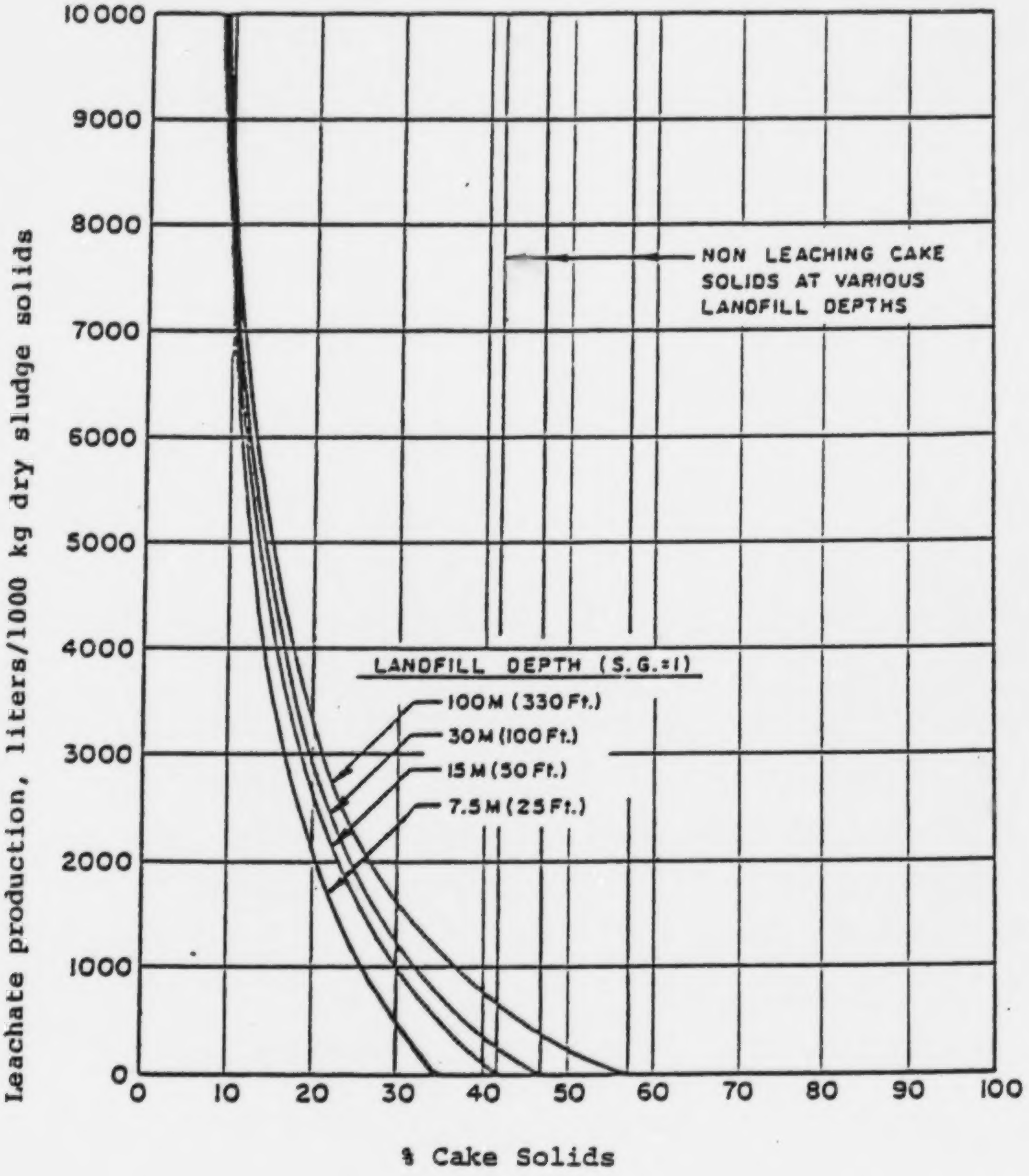

Figure 6. Leachate production resulting from sludge compaction (Parkhurst, 1978).

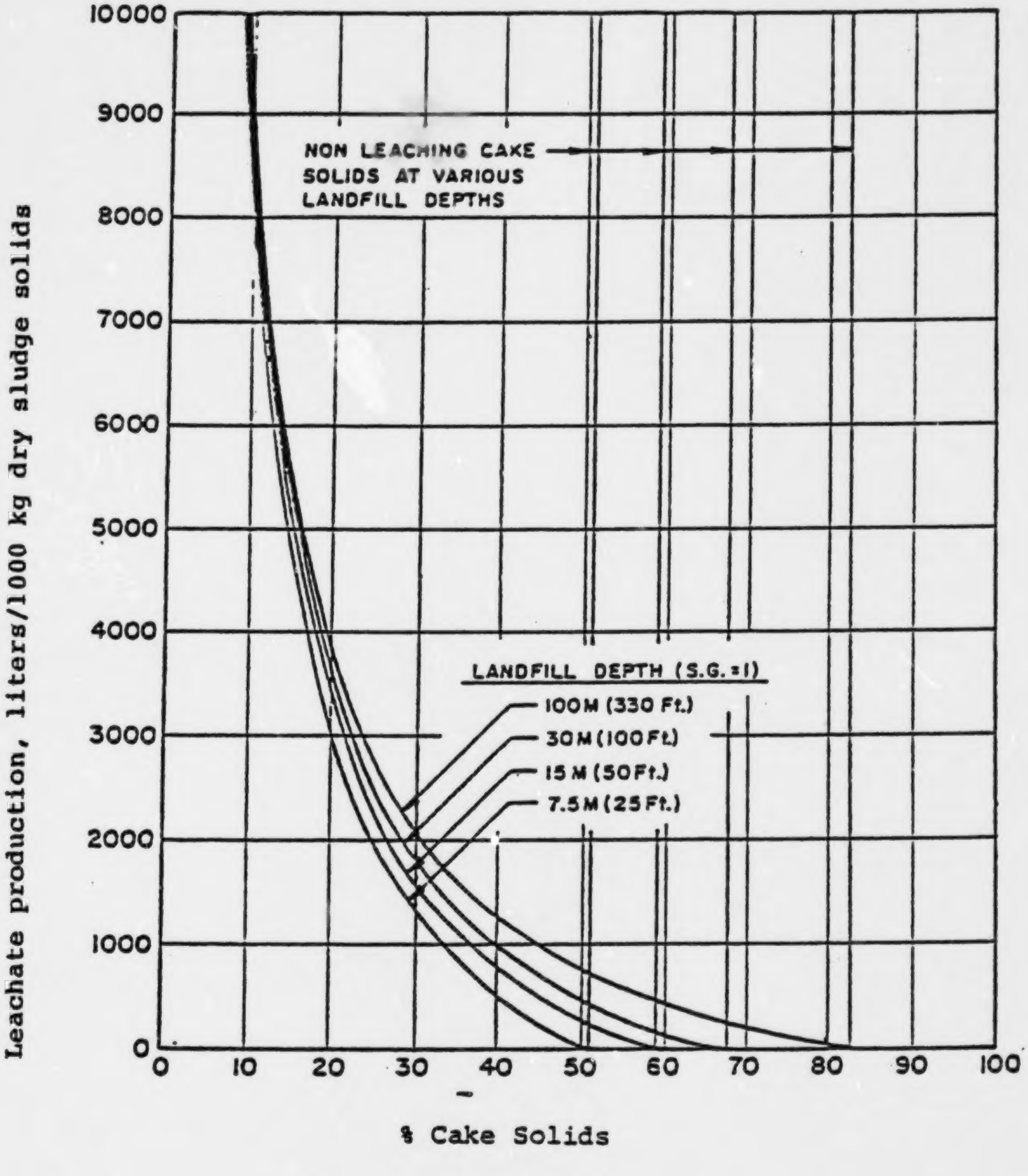

Figure 7. Leachate production resulting from sludge compaction and decomposition (Parkhurst, 1978).

Evapotranspiration

Evapotranspiration is the sum of water loss by evaporation and transpiration (plant water consumption). Factors controlling evaporation and transpiration are well recognized, but quantitative evaluation is difficult because of their interdependent effects (Chow, 1964). For a free-water surface, the rate of evaporation depends on the vapor pressure of the body of water and that of the air. Major factors affecting these vapor pressures include temperature (both water and air), wind, humidity, atmospheric pressure, quality of the water, and the nature and shape of the surface. With the exception of the last two, the factors vary seasonally and geographically. Average annual pan evaporation presented in Figure 8 illustrates geographical variations. From this figure, the mean annual net effective precipitation, which is the difference between "effective precipitation" (Figure 4) and "potential evaporation (Figure 8), for the U. S. can be prepared (Figure 9). Negative values of net effective precipitation represent potential water loss from the land surface to the atmosphere (i.e., evaporation > effective precipitation). Landfills located in areas with negative values of net effective precipitation are expected to have a negligible source of leaching water from precipitation.

Evaporation is proportional to the vapor pressure difference between water in the soil and the air, so temperature increase may not proportionately affect the evaporation rate and thus, a high correlation between the air temperature and evaporation cannot be expected. Wind removes water molecules from the region near the surface, thus also increasing evaporation rates. Humidity and atmospheric pressure are inversely related to evaporation. Evaporation rates decrease as dissolved solids concentrations increase. The evaporation rate decreases about 1 percent for each 1 percent increase in specific gravity (Keen, et al., 1926; Fisher, 1927; Penman, 1948; Veihmeyer, et al., 1955; Chow, 1962).

Evaporation of water from soil surfaces is controlled by the same factors that evaporate water from free-water surface. A difference is that soil particles tend to bind water molecules against evaporation (Penman, 1948). This attractive force is a function of the moisture content of the soil and the characteristics of the soil. Unsaturated wet soils maintain a nearly constant rate of evaporation over a range of moisture content (Keen, et al., 1926; Fisher, 1927). Evaporation is reduced, according to Veihmeyer, et al., (1955), at or below the permanent wilting point* of the soil. Evaporation from the soil surface

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* Permanent wilting point, or permanent wilting percentage, is the moisture content of soil at which soil cannot supply water at a sufficient rate to maintain turgor, resulting in wilting of vegetation.

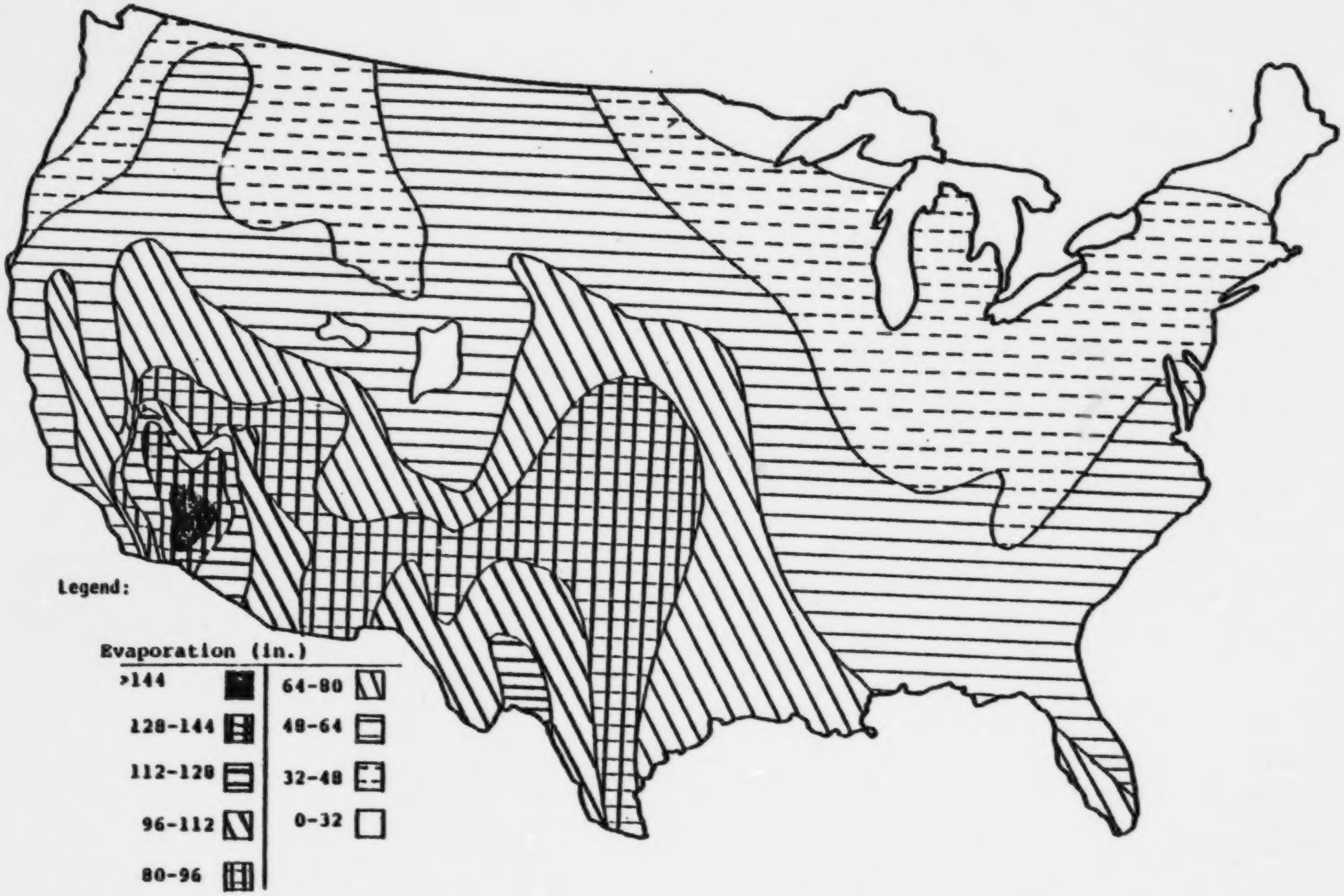
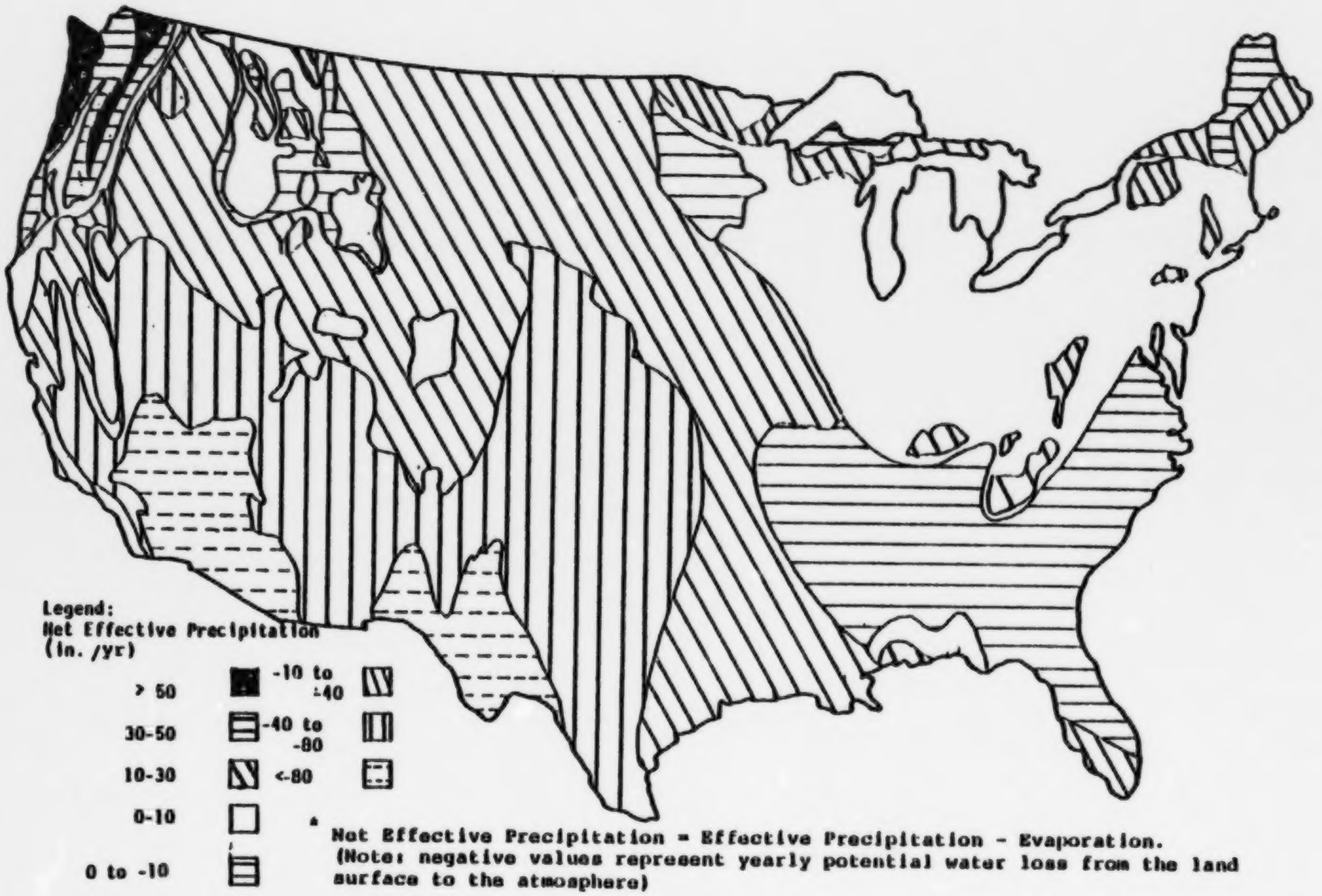

Legend:
Evaporation (in.)
>144
128-144
112-128
96-112
80-96
64-80
48-64
32-48
0-32

Figure 8. Mean annual pan evaporation (after Chow, 1964).


Legend:
Net Effective Precipitation
(in. /yr)
> 50
30-50
10-30
0-10
0 to -10
-10 to -40
-40 to -80
<-80

* Net Effective Precipitation = Effective Precipitation - Evaporation.
(Note: negative values represent yearly potential water loss from the land surface to the atmosphere)

Figure 9. Mean annual net effective precipitation (prepared from Figures 4 and 8).

will continue as long as the shallow surface layer (about 100 mm (4 in.) for clays and about 200 mm (8 in.) for sands) contains moisture above the permanent wilting point. Evaporation from deeper soil is insignificant (Chow, 1964).

Factors affecting transpiration are physiological and environmental. Physiological factors include the type and density of vegetation, leaf structure, and plant condition and age. Table 1 provides examples of seasonal transpiration ranges for several types of vegetation (Urquhart, 1959). The data shows transpiration has the potential to remove significant fractions of water from the soil.

TABLE 1. APPROXIMATE SEASONAL TRANSPIRATION OF SELECTED VEGETATION (URQUHART, 1959).

Vegetation	Transpiration (in./growing season)
Alfalfa and Clover	>2.5
Coniferous Trees	4-9
Deciduous Trees	7-10
Lucern Grass	26-65
Meadow Grass	22-60
Oats	28-40
Rye	>18
Wheat	20-22

Among the environmental factors influencing transpiration are season, temperature, solar radiation, relative humidity, wind speed, and soil moisture when the permanent wilting point is reached (Caffrey, et al., 1974; Molz, et al., 1974). The season of the year and the associated sunlight conditions will affect the temperature of the leaf and thus, transpiration. Solar radiation is particularly important because it stimulates the "guard cells"* to open the leaf pores. About 95 percent of daily transpiration occurs between sunrise and sunset (Molz, et al., 1974). The removal of water vapor next to the leaf surface by wind also can increase transpiration. With wind speeds of eight to twenty-four kilometers per hour (5 to 15 mph), transpiration rates may be twenty and fifty percent higher than at zero wind speed (Chow, 1964; Molz, et al., 1974). Transpiration is also affected by the soil moisture content when it is reduced to the permanent wilting point (Chow, 1964).

Evaporation and transpiration are difficult to measure separately. Consequently, water lost to the atmosphere is most often reported as total evapotranspiration. Methods of calculating evapotranspiration are presented in the next subsection.

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* Pores on the leaf surfaces that are surrounded by special cells are called "guard cells."

Infiltration

Infiltration is the flow of water through the soil surface. It is only one of the possible fates for water applied to the landfill surface, the other possible fates being runoff and evapotranspiration. Factors affecting the rates of surface runoff and evapotranspiration will, therefore, indirectly affect infiltration. The characteristics of the cover material (permeability, moisture content, porosity, organic content, degree of compaction, thickness of surface permeable layers, etc.), surface topography (slope, surface storage capacity, etc.), vegetation, and underdrainage condition were found to influence the infiltration directly (Chow, 1964; Remson, et al., 1968; Gray, et al., 1968; Caffrey, et al., 1974; Viessman, et al., 1977; and Lutton, et al., 1979).

The most important factor affecting infiltration is usually the moisture content of the landfill cover material. Infiltration rates are reduced with an increase in the moisture content in the top soil (Chow, 1964; Gray, et al., 1968). This relationship can be illustrated by the infiltration rate changes after a rainstorm, as shown in Figure 10. The maximum infiltration rate occurs at the beginning of the rainstorm when the soil is relatively dry. The rate decreases first rapidly and then slowly approaching a stable minimum. the minimum infiltration rate usually approaches the percolation rate (to be discussed later) of the soil. The changes of infiltration rates versus time during a rainstorm can be approximated by the following equation (Viessman, et al., 1977):

$$f = fc + (fo-fc)\ e^{-kt} \qquad (3)$$

where:

f = infiltration rate at some time t;

k = a constant representing the rate of decrease in f;

fc = a final or equilibrium rate; and

fo = the initial infiltration rate.

Infiltration into a landfill surface is also determined by the structure of the surface layer of soil. Important characteristics of soil tending to retard infiltration are high non-capillary porosity* and high organic content of the subsoil

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* Non-capillary porosity is the fraction of air-filled voids in a porous medium at field capacity.

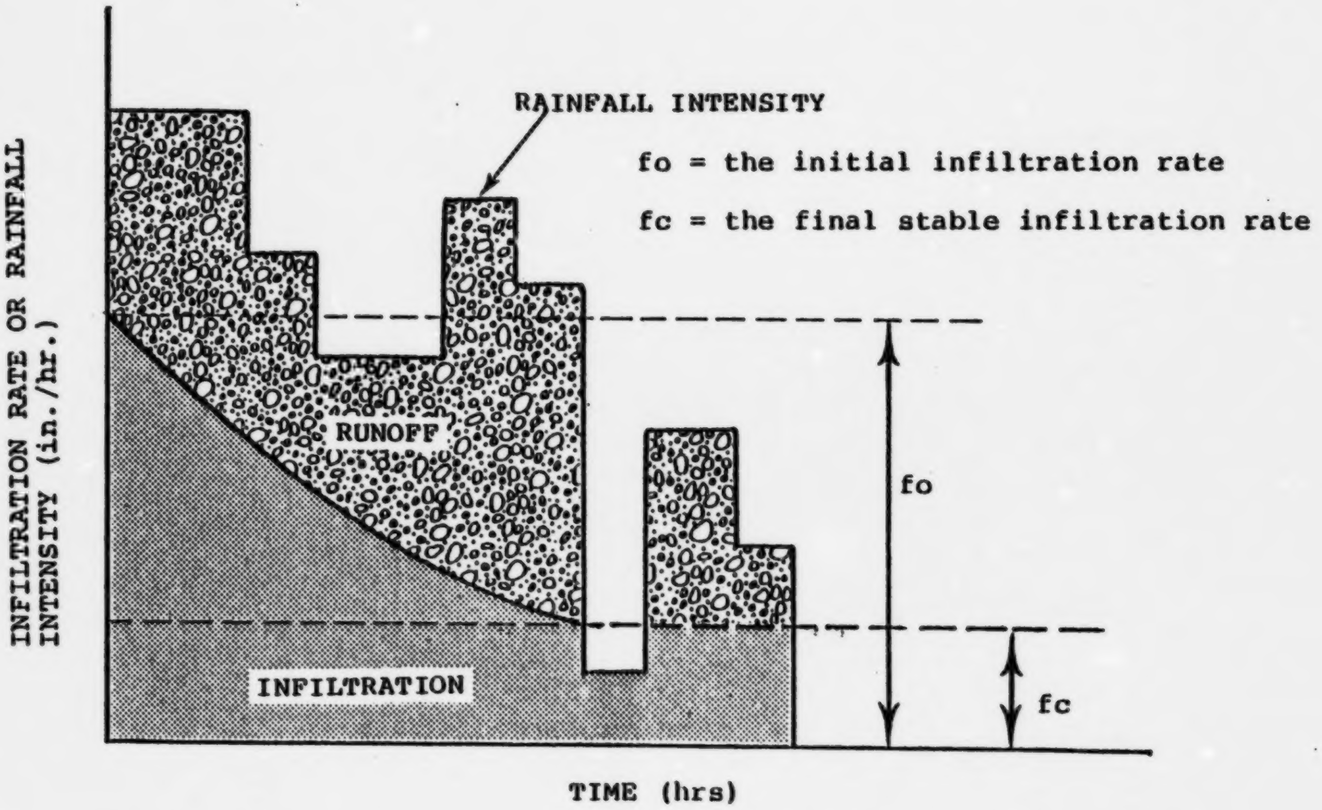

RAINFALL INTENSITY
fo = the initial infiltration rate
fc = the final stable infiltration rate
INFILTRATION RATE OR RAINFALL INTENSITY (in./hr.)
RUNOFF
INFILTRATION
fo
fc
TIME (hrs)

Figure 10. Relationship of rainfall, runoff, and infiltration during a rainstorm event.

(Chow, 1964). Raindrops falling on bare soil tend to compact the surface layer and wash fines into the large pores, also causing a sharp reduction in infiltration.

The type and density of landfill vegetation are other important factors affecting infiltration. Vegetation protects the soil surface from the impact of rainfall. It also retards surface flow velocity so that water is retained on the landfill surface for a longer time.

Water can enter the soil no more rapidly than it flows downward within the soil. Underdrainage, therefore, affects infiltration. Where underlying layers are unsaturated, the infiltration rate is regulated by the permeability of the upper layers. Where underlying layers are saturated, the layer with least permeability will control infiltration (Chow, 1964).

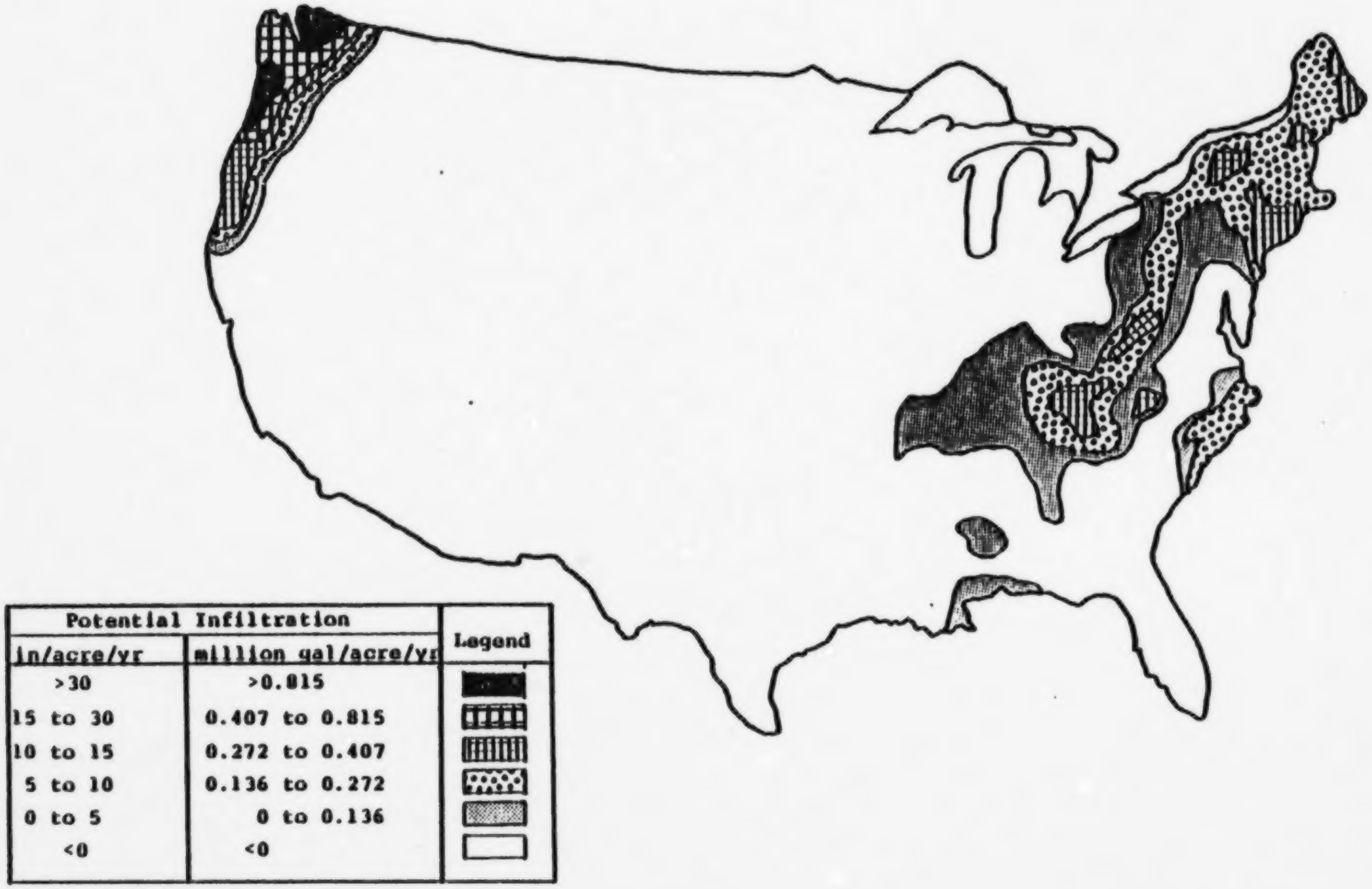
Figure 11 is constructed to compare the potential infiltration for landfills located in different goegraphical areas of the United States. This figure is derived from the mean annual net effective precipitation (Figure 9) and the potential landfill surface runoff (0.25 is assumed for the runoff coefficient). This figure is not to be misconstrued as showing the actual amount of water that infiltrates, but rather provides a comparison of potential landfill infitration rates. Such potential exists for the northwestern, northeastern, and mid-eastern United States. However, this does not mean that leachate will not be a problem from precipitation in other areas. Leachate could very easily be generated during a deluge-type rainstorm lasting hours or a prolonged rainfall over a period of days.

Moisture Retention

Figure 12 depicts the moisture retention properties of an unsaturated soil or refuse (Bauer, 1956). Commencing with saturation, the water initially drains rapidly and the rate progressively decreases. Field capacity remains essentially unchanged unless the vegetation root zone is in the layer of consideration. A soil or refuse located within the root zone can be dried to the permanent wilting point as shown in Figure 12. Approximate field capacities and wilting points for soil and refuse are discussed in the next subsection.

Percolation

Landfill percolate is generally defined as the quantity of water which exceeds the refuse field capacity. Theoretically, the water movement through a compacted refuse cell will not occur until the refuse field capacity is exceeded. Practically, because of the heterogeneous nature of the refuse, some channeling of water may occur causing some percolation to appear prior to attainment of field capacity. In field situations, the



Potential Infiltration		Legend
in/acre/yr	million gal/acre/yr	
>30	>0.815	■
15 to 30	0.407 to 0.815	▦
10 to 15	0.272 to 0.407	▥
5 to 10	0.136 to 0.272	⠿
0 to 5	0 to 0.136	▒
<0	<0	□

Figure 11. Potential infiltration for landfills in various geographical areas of the United States (See text for explanation).

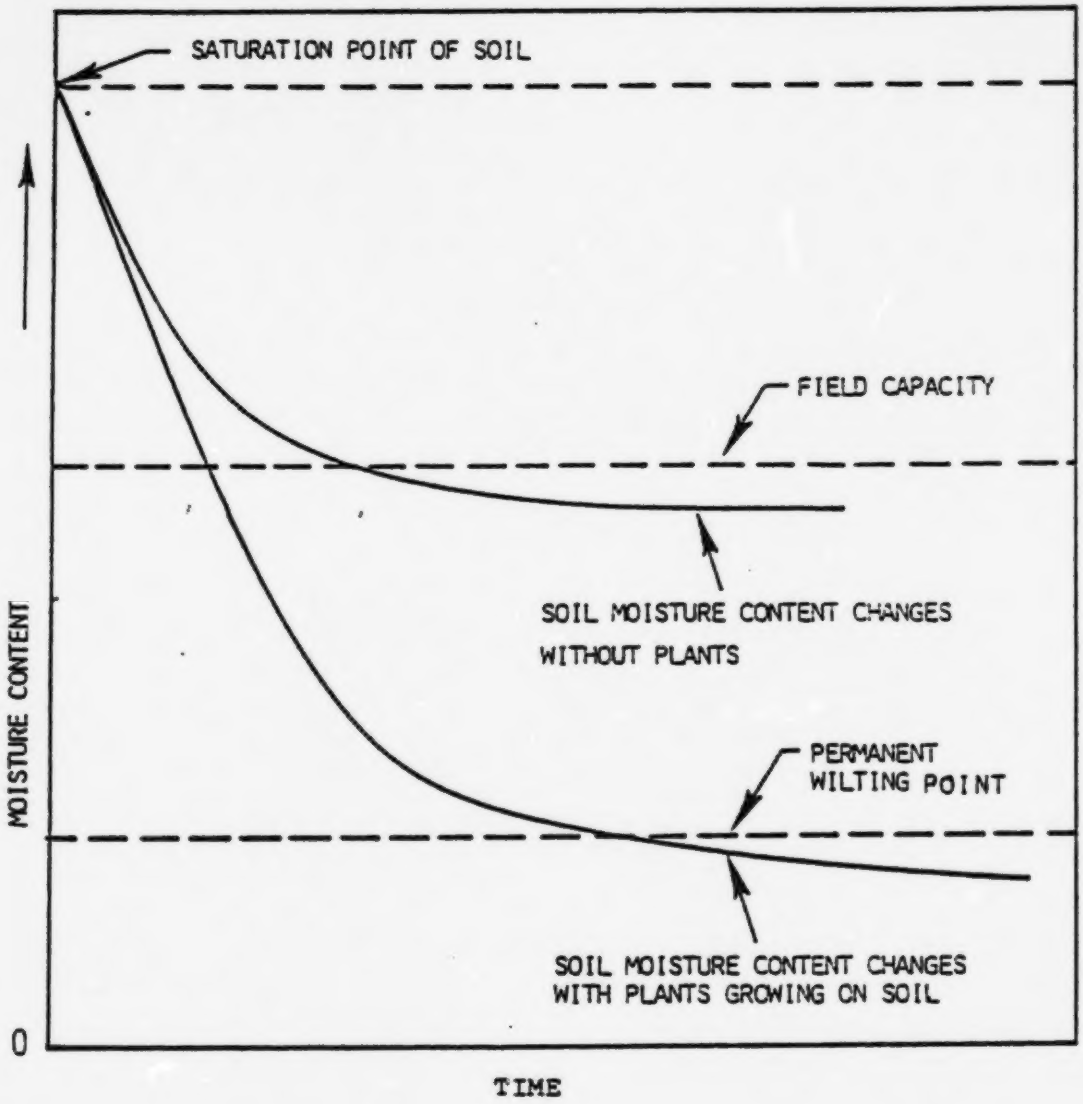


Figure 12. Moisture content as a function of time for a draining soil or refuse (Bauer, 1956).

significance of such a channeling effect is difficult to estimate. Due to the fact that the refuse seldom achieves saturation (e.g., Remson, et al., 1968; Rover, et al., 1973; and Lutton, et al., 1979), and the high water absorbing capability (to be discussed later) of most refuse, the channeling amount may be small. Fenn, et al. (1975) suggested that the channeling amount should be small and certainly not a continuous flow, and was assumed negligible in their percolation estimations.

As water percolates vertically through a landfill soil cover, and into the refuse, layers with different structures and properties are encountered. If water is moved in a saturated layer of the refuse (which may occur for top layers of the refuse right after a deluge-type or prolonger rainstorm), the gravity and pressure forces, which form the hydraulic gradient, will influence the water movement. The velocity and flow rate of such percolates can be expressed by Darcy's Law as follows:

$$V = Ki \qquad (4)$$

and $$Q = KAi \qquad (5)$$

where:
V = flow velocity;
Q = flow rate;
K = refuse permeability;
A = cross-sectional area perpendicular to the flow direction; and
i = hydraulic gradient.

The refuse permeability as shown above is one of the most important parameters for determining percolation rates in the saturation condition. It is generally concluded that the permeability of refuse is affected by refuse particle size and density. However, the quantitative relationship is greatly lacking. A recent study conducted by Fungaroli, et al. (1979) could not establish a significant relationship between saturated permeability, density and milled refuse size. Nevertheless, the ranges of refuse permeability was available and reported to be 10^{-2} cm/sec for the low density (less than 500 pounds per cubic yard) large particle refuse, to 10^{-4} cm/sec for the high density (greater than 500 pounds per cubic yard) large and small particle refuse (Fungaroli, et al., 1979).

Water movement in an unsaturated layer of the refuse fill is affected by thermo-osmosis and capillary forces (Chow, 1964; De Wiest, 1967, in addition to gravity and pressure forces, thermo-osmosis forces can move water by diffusion of water vapor in the soil or refuse pores. This phenomenon may be significant especially when warm temperatures exist resulting in water vaporization (De Wiest, 1967). In an active landfill, the thermophilic condition could result in such an effect. However, a study on the thermo-osmosis effect for water movement in the unsaturated landfill is lacking. Chow (1964) stated that the capillary potential gradients are large and usually control the water movement throughout most of the moisture content ranges of the unsaturated layer. The modified Darcy's Law for unsaturated flow was thus derived, mainly based on gravity and capillary potential (Chow, 1964; and Philip, 1969). In this derivation, the permeability, K, in Equations (4) and (5) becomes a nonlinear function of moisture content, Θ. Figure 13 is an example of such a nonlinear relationship.

Darcy's Law for unsaturated flow indicates that any moisture movement must occur in the direction of decreasing gravity and capillary potential (Chow, 1964). The gravity potential is constant in a landfill. In order for a wetting front to move downward from a fine medium into a coarse medium in an unsaturated refuse, the fine medium must become nearly saturated to create sufficient capillary potential (Colman, et al., 1944). In the case of flow from a coarse medium into an underlying finer stratum, the rate of percolation is controlled exclusively by the fine particles because of the relatively low permeability of the fine particles. Where water is applied more rapidly than the fine-grained percolation capacity, either the upper layer will become saturated, or the water will flow laterally.

After the percolate reaches the landfill/underlying soil boundary, leachate will form. Percolate is identical with leachate generated if ground-water intrusion, changes in moisture retention, and biochemical contributions are negligible (Fenn, et al., 1977; Lutton, et al., 1979). Percolation can be estimated by direct measurement or calculation. Percolation is measured with a sub-surface water trap. Water traps and their application are discussed in the literature (Fenn, et al., 1977). The calculation methods for percolation are discussed in the next subsection.

CONCEPTS AND TECHNIQUES DESCRIBING LEACHATE GENERATION

Numerous mathematical methods have been used for quantitative estimation of the volume of leachate generated from landfills (Thornthwaite, et al., 1957; Remson, et al., 1968; Fenn, et al., 1975; Dass, et al., 1979; Duvel, et al., 1979; and Lutton, et al., 1979). Although the approaches employed in these methods vary, they are all basically derived from the water balance or water budget principle. The water balance principle is based on: (1) a one-dimensional flow model and conservation of mass relationships among various components of the leachate sources, and (2) the retention and transmission characteristics of the refuse and cover soil. The generalized water balance relationships are shown in Table 2 and are illustrated in Figure 14.

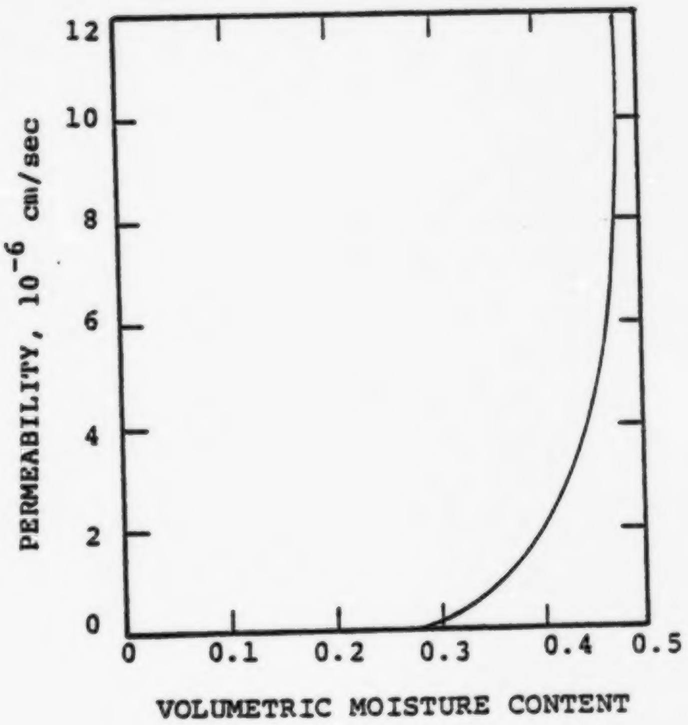


Figure 13. Example showing the effect of moisture content on permeability for a clay soil (after Philip, 1969).

TABLE 2. GENERALIZED WATER BALANCE EQUATION AT A MUNICIPAL LANDFILL SITE

$W_P + W_{SR} + W_{IR} = I + R$ ---------------------------------- (6)

where:

W_P = Input water from precipitation

W_{SR} = Input water from surrounding surface runoff

W_{IR} = Input water from irrigation

I = Infiltration

R = Surface runoff

$PER_S = I - E - \Delta S_S$ -- (7)

$PER_R = I - E - \Delta S_S + W_D - \Delta S_R$ --------------------------- (8)

$= PER_S + W_D - \Delta S_R$ ---------------------------------- (9)

where:

PER_s and PER_R = Percolation in soil and refuse respectively

W_D = Water contributed by solid waste decomposition

ΔS_S = Change in moisture storage in soil

ΔS_R = Change in moisutre storage in refuse

E = Evapotranspiration

and $L = PER_R + W_{GW}$ ----------------------------- (10)

where:

L = Leachate generation

W_{GW} = Input water from underflow

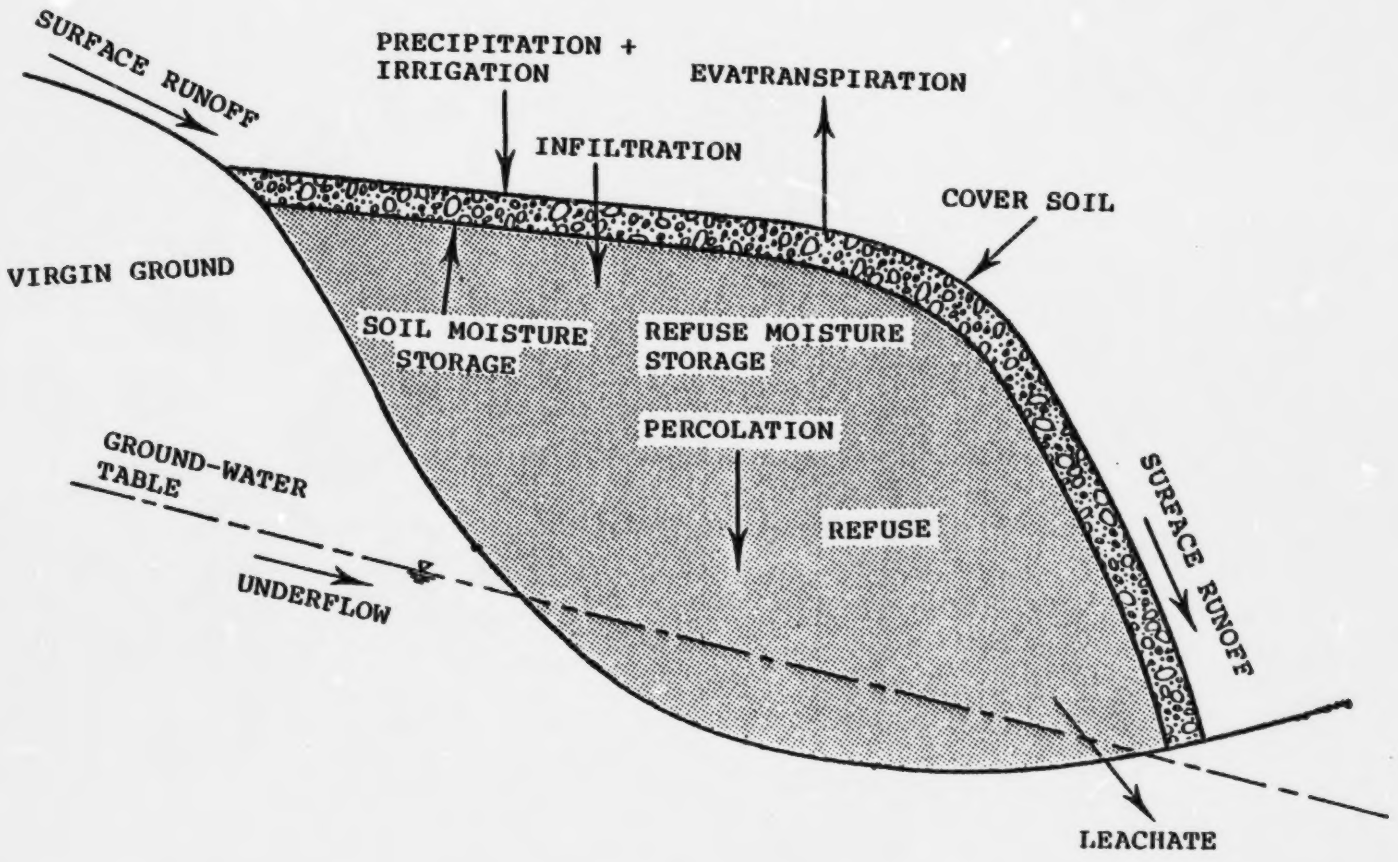

SURFACE RUNOFF
PRECIPITATION +
IRRIGATION
EVATRANSPIRATION
INFILTRATION
COVER SOIL
VIRGIN GROUND
SOIL MOISTURE
STORAGE
REFUSE MOISTURE
STORAGE
PERCOLATION
GROUND-WATER
TABLE
REFUSE
SURFACE RUNOFF
UNDERFLOW
LEACHATE

Figure 14. Municipal landfill water balance.

Equation (6) in Table 2 illustrates the water balance relationship of the landfill surface. Water contributed by precipitation (W_p), surface runoff (W_{SR}), or irrigation (W_{IR}) will either become surface runoff (R), or infiltrate into the cover soil (I). A portion of the infiltrated water leaves by evapotranspiration (E) and part will recharge the cover soil. Once the field capacity of the cover soil is attained, vertical percolation (PER_S) will occur. The quantity of percolation through the cover material can be calculated by using equation (7). At first the percolate from the cover material will be absorbed by the refuse. As the landfill refuse reaches field capacity, refuse percolate (PER_R) is generated in quantities described by equation (8) or (9). The refuse percolate will eventually evolve as refuse leachate (L). If ground-water intrustion (W_{GW}) occurs, leachate generation estimates can be modified using equation (10) (Table 2).

Because leachate generation mechanisms are site specific, any estimation method requires knowledge of site topography, geology, hydrology, climatology, and meteorology. The many poorly known factors in the water balance equations often require empirical, rational, or experimental methods. These different approaches reflect the wide variety of concepts describing leachate generation. In this report, only those techniques which are appropriate for municipal landfills are discussed.

Determining Precipitation

The amount of precipitation is expressed as the amount of water (in inches or millimeters) which would accumulate on a sealed level surface. For the purposes of leachate quantity estimation, existing precipitation records or gauge measurements may be used to determine precipitation quantities.

Existing Precipitation Records--

The U.S. Weather Bureau collects precipitation data for the U.S. The methods of handling the data and its publication have varied greatly since the inception of the Bureau in 1891. Records are presently stored at the National Climatic Center at Asheville, North Carolina.[*] Local, regional or national precipitation data can be obtained from this agency.

Several states and local agencies publish data from rain gauges maintained by them. Precipitation data may also be obtained from military installations, local airports, harbors, fire stations, and universities. Independent data maintained by these organizations are sometimes very useful for specific landfill site locations.

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[*] National Climatic Center, National Oceanic and Atmospheric Administration, Asheville, North Carolina 28801, (704) 258-2850.

Precipitation Gauges--

Precipitation gauges can be installed on or near a landfill for extended leachate generation investigations. Precipitation gauges may be of the nonrecording or recording type. The standard U.S. Weather Bureau nonrecording gauge is a circular cylinder 20 cm (8 in. in diameter), which serves as an overflow can. A funnel shaped receiver of the same diameter is connected to a measuring tube with a cross section equal to one-tenth that of the receiver. When 2.5 cm (1 in.) of precipitation falls into the receiver, the measuring tube is filled to a depth of 25 cm (10 in.). Precipitation is measured with a graduated rod to the nearest hundredth of a centimeter or hundredth of an inch. Many commercially available gauges exist, and the selection of one should be made with the particular purpose of the monitoring program in mind.

Recording gauges are also available. The weighing gauge and the tipping-bucket gauge are two common types. The weighing gauge has a bucket supported by a spring or lever balance. Movement of the bucket is transmitted to a pen which traces a record of the increasing weight on a clock-driven chart or punched paper tape. The tipping-bucket gauge consists of a pair of buckets pivoted under a funnel in such a way that when one bucket receives 0.01 cm (or 0.01 in. depending on equipment used) of precipitation, it tips and discharges its contents into a reservoir. The other bucket is then positioned under the funnel. A recording mechanism indicates the time of occurrence of each tip. The tipping-bucket gauge is well adapted to the measurement of rainfall intensity for short periods. The weighing gauge is more rugged and can record snowfall.

Determining Surface Runoff

Surface runoff on or adjacent to a landfill site can be determined by surface measurement, empirical formulas, or graphic methods. Exact measurement is a difficult task because of the non-point discharge characteristics of the overland flow and the large area involved. Nevertheless, actual runoff measurements may be obtained by carefully fencing a test plot of the landfill or a test section of the proposed cover system. The rational equation, one of the empirical formulas, is widely used by hydrologists and sanitary engineering designers (Chow, 1964; and American Society of Civil Engineers, 1960). Although this formula is based on a number of assumptions which cannot be readily satisfied under actual circumstances, its simplicity makes it a practical tool. Many other empirical formulas or graphical solutions for surface runoff have been suggested by researchers. Hundreds have become almost obsolete in design practice because of their complexity or because the values of coefficients in the formulas are not known in terms of specific conditions (Chow, 1964). The following section will discuss only those methods suitable for landfill situations.

Surface Measurement--

Landfill surface runoff caused by precipitation or irrigation can be directly measured either in the field or in laboratory pilot studies. For field measurements, fencing a test plot of the landfill will collect the overland flow. If tests are performed on certain sections of the landfill, representative vegetation, slopes, soil types, and other landfill surface conditions should be considered. A precipitation gauge should be located nearby to record precipitation daily or even more frequently.

Rational Method--

The rational method is usually expressed in terms of the following equation (Chow, 1964):

$$R = C \cdot P \cdot A \text{--} (11)$$

where R and P are surface runoff peak discharge and uniform rate of rainfall intensity, respectively. A is the landfill area. The runoff coefficient, C, indicates the fraction of total precipitation that flows off the surface. If the units for P and A are inches per hour and acres, then the unit for R is in cfs. Equation (11) can also be written:

$$R = C \cdot W_p \text{--} (12)$$

where W_p is the input precipitation in inches or mm. In this case, the units for R become in. or mm depending on the units of W_p.

The key to the successful estimation of surface runoff by the rational method is the correct choice of the runoff coefficient. The runoff coefficient for a particular area depends on surface characteristics, type and extent of vegetation, surface slope, and other less important factors as described previously. Considerable work has been done developing more or less standard runoff coefficients for various agricultural and engineering situations. However, runoff coefficients for municipal landfills have received less attention. Some values of the runoff coefficient were reported by a joint committee of the American Society of Civil Engineers and the Water Pollution Control Federation (Table 3) (American Society of Civil Engineers, 1960). These values are for storms of 5- to 10-year frequencies. Values shown in Table 4 are based on individual effects of topography, soil type and cover conditions (Bernard, 1932). The "Engineering Manual" edited by Perry, suggests a set of runoff coefficients based on cover material and slope (Table 5) (Perry, 1976). Salvato, et al. (1971) in their landfill leachate study also suggested a set of runoff coefficients

(Table 6). Because there are so many physiographic and physical variables that can affect the runoff coefficient, the results obtained are not consistent. Additional investigation of the runoff coefficients for landfill sites is required if rational estimation methods are to be used. Significant information omitted in the above tables include: the quantitative relationships between runoff and precipitation duration, precipitation frequency, antecedent soil moisture content, and the degree of compaction of the cover material.

TABLE 3. RUNOFF COEFFICIENTS FOR STORMS OF 5- TO 10-YEAR FREQUENCIES (American Society of Civil Engineers, 1960).

Surface Condition	Runoff Coefficient, C
Grass Cover:	
Sandy soil, flat, 2%	0.05 - 0.10
Sandy soil, average, 2-7%	0.10 - 0.15
Sandy soil, steep, 7%	0.15 - 0.20
Heavy soil, flat, 2%	0.13 - 0.17
Heavy soil, average, 2-7%	0.18 - 0.22
Heavy soil, steep, 7%	0.25 - 0.35
Suburban Area	0.25 - 0.40
Playground	0.20 - 0.35
Unimproved Area	0.10 - 0.30
Drives and Walks	0.75 - 0.85

Curve Number Method--

The curve number method is suggested by the U.S. Department of Agriculture, Soil Conservation Service (SCS) for predicting direct surface runoff on agricultural land (SCS, 1972). Estimation of runoff is based on the amount of rainfall, soil type, land use, land cover, and antecedent moisture condition. Calculation procedures are as follows:

(1) Determine the antecedent moisture condition--
The antecedent moisture conditions of cover soils are classified into three groups, based on 5-day total antecedent rainfall: AMC I,

TABLE 4. RUNOFF COEFFICIENTS FOR DRAINAGE AREAS WITH DIFFERENT TOPOGRAPHY, SOIL, AND COVER CONDITIONS* (Chow, 1964; and Bernard, 1932)

Site Description	Value, v'
Topography:	
Flat land, with average slope of 1 to 3 ft/mile	0.30
Rolling land, with average slope of 15 to 20 ft/mile	0.20
Hilly land, with average slope of 150 tp 250 ft/mile	0.10
Soil:	
Tight impervious clay	0.10
Medium combinations of clay and loam	0.20
Open sandy loam	0.40
Cover:	
Cultivated lands	0.10
Pasture	0.15
Woodland	0.20

---

* Add values v' for topography, soil and cover, and subtract from unity to obtain runoff coefficient.

TABLE 5. RUNOFF COEFFICIENTS AS AFFECTED BY COVER MATERIAL AND SLOPE (Perry, 1976)

Type of area	Runoff Coefficient, c		
	Flat: slope <2%	Rolling: slope 2-10%	Hilly: slope >10%
Grassed Areas	0.25	0.30	0.30
Earth Areas	0.60	0.65	0.70
Meadows and Pasture Lands	0.25	0.30	0.35
Cultivated Land:			
Impermeable (clay)	0.50	0.55	0.60
Permeable (loam)	0.25	0.30	0.35

TABLE 6. RUNOFF COEFFICIENTS USED BY SALVATO, ET AL. FOR LEACHATE ESTIMATION (Salvato, et al., 1971).

Surface Condition	Percent Slope	Percent Surface Runoff		
		Sandy Loam	Clay or Silt Loam	Clay
Pasture or Meadow	0-5 (flat)	10	30	40
(Surface with cover crop)	5-10 (rolling)	16	36	55
	10-30 (hilly)	22	42	60
No Vegetation	0-5 (flat)	30	50	60
(Raw Soil Surface)	5-10 (rolling)	40	60	70
	10-30 (hilly)	52	72	82

AMC II, and AMC III. Table 7 provides criteria for these three classifications.

(2) Determine the hydrologic soil group--
The hydrologic properties of a soil or a group of soils can be classified according to their properties as described in Table 8.

(3) Determine the land use or cover and the hydrologic conditions--
Land cover is any material (but usually vegetation) covering the soil and providing protection from the impact of rainfall. Detailed information about the land cover is seldom available. It is therefore necessary to rely on land use as an index of cover conditions. The types of land use or cover are listed in Table 9. The hydrologic condition of the vegetation should also be determined for runoff estimation. Good, fair, and poor conditions are designated for indicating vegetation density. Good conditions are represented by a high proportion of alfalfa or other close-seeded legumes or grasses that will improve tilth and increase infiltration. Poor conditions occur for raw crops, small grains, and fallow land in various combinations. A vegetated sanitary landfill cover will be similar to a pasture, range, or meadow. The classifications of hydrologic conditions can be seen in Table 9.

TABLE 7. RAINFALL LIMITS FOR ESTIMATING ANTECEDENT MOISTURE CONDITIONS (Soil Conservation Service, 1972)

Antecedent Moisture Condition Class	5-day Total Antecedent Rainfall, in.*	
	Dormant Season	Growing Season
I	Less than 0.5	Less than 1.4
II	0.5 to 1.1	1.4 to 2.1
III	Over 1.1	Over 2.1

* 1 in. = 2.54 cm.

(4) Determine the value of curve number (CN)--
After the antecedent moisture, hydrologic soil and cover conditions are determined, the value of the curve number (CN) can be determined from the SCS curve number table. Table 9 provides runoff curve numbers for various combinations of soils and

TABLE 8. HYDROLOGIC SOIL GROUPS USED BY THE SOIL CONSERVATION SERVICE (Soil Conservation Service, 1972)

Hydrologic Soil Group	Soils Included*
A	(Low runoff potential) Soils having high infiltration rates even when thoroughly wetted, consisting chiefly of sands or gravel that are deep and well to excessively drained. These soils have a high rate of water transmission.
B	Soils having moderate infiltration rates when thoroughly wetted, chiefly moderately deep to deep, moderately well to well drained, with moderately fine to moderately coarse textures. These soils have a moderate rate of water transmission.
C	Soils having slow infiltration rates when thoroughly wetted, chiefly with a layer that impedes the downward movement of water or of moderately fine to fine texture and a slow infiltration rate. These soils have a slow rate of water transmission.
D	(High runoff potential) Soils having very slow infiltration rates when thoroughly wetted, chiefly clay soils with a high swelling potential; soils with a high permanent water table; soils with a clay pan or clay layer at or near the surface; and shallow soils over nearly impervious materials. These soils have a very slow rate of water transmission.

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* Soils are classed in the next lowest category when a high percentage of stones is present.

TABLE 9. RUNOFF CURVE NUMBERS FOR HYDROLOGIC SOIL-COVER COMPLEXES (FOR AMC II)[*]

Land Use or Cover	Treatment or Practice	Hydrologic Condition	Hydrologic Soil Group			
			A	B	C	D
Fallow	Straight row	Poor	77	86	91	94
Row crops	Straight row	Poor	72	81	88	91
	Straight row	Good	67	78	85	89
	Contoured	Poor	70	79	84	88
	Contoured	Good	65	75	82	86
	Contoured & terraced	Poor	66	74	80	82
	Countoured & terraced	Good	62	71	78	81
Small grain	Straight row	Poor	65	76	84	88
	Straight row	Good	63	75	83	87
	Contoured	Poor	63	74	82	85
	Contoured	Good	61	73	81	84
	Contoured & terraced	Poor	61	72	79	82
	Contoured & terraced	Good	59	70	78	81
Close-seeded legumes[+] of rotation meadow	Straight row	Poor	66	77	85	89
	Straight row	Good	58	72	81	85
	Contoured	Poor	64	75	83	85
	Contoured	Good	55	69	78	83
	Contoured & terraced	Poor	63	73	80	83
	Contoured & terraced	Good	51	67	76	80
Pasture or range[#]		Poor	68	79	86	89
		Fair	49	69	79	84
		Good	39	61	74	80
	Contoured	Poor	47	67	81	88
	Contoured	Fair	25	59	75	83
	Contoured	Good	6	35	70	79
Meadow (permanent)		Good	30	58	71	78
Woodlands (farm woodlots)		Poor	45	66	77	83
		Fair	36	60	73	79
		Good	25	55	70	77
Farmsteads			59	74	82	86
Roads, dirt[‡]			72	82	87	89
Roads, hard-surface[‡]			74	84	90	92

[*] Reference: Soil Conservation Service (1972)

[+] Close-frilled or broadcast

[‡] Including right-of-ways

[#] Classification of native pasture or range:

Hydrologic Condition	Vegetative Condition
Poor	Heavily grazed, no mulch, or having plant cover on less than about 50 percent of the area.
Fair	Moderately grazed; between about 50 and 75% of the area with plant cover.
Good	Lightly grazed; more than about 75% of the area with plant cover.

covers at AMC II. The CN values of other AMC's can be found by using a conversion table shown in Table 10.

(5) Estimate direct runoff--
Direct runoff can be estimated by a graph (Figure 15). Figure 15 is appropriate for use with storms of 1-day duration or less. The time of day during which the runoff occurred must be found in detailed rainfall records. When a storm occurs over several days, the runoff estimate may be made on a daily basis, changing the antecedent mositure condition accordingly. The total runoff for a certain period also can be estimated directly, using the value of CN applicable for the first day of the storm, and the total storm rainfall in the period. In some cases such an estimate will differ slightly from the one made on a day-by-day basis. The direct runoff estimation also can be calculated by the following equation:

$$R = \frac{\left[W_p - 0.2\left(\frac{1000}{CN} - 10\right)\right]^2}{W_p + 0.8\left(\frac{1000}{CN} - 10\right)} \text{ ----------(13)}$$

where R, WP are direct runoff and rainfall respectively (unit in inches).

Cook's Method--

Cook's method uses an empirical relationship between drainage area and peak flow with modifications for climate, relief, infiltration, vegetal cover, and surface storage (Chow, 1964). The summation of applicable values of W (Table 11) reflects the hydrologic conditions of the watershed. Figure 16 illustrates the relationship between drainage area and peak flow for various hydrologic condtions. Values obtained (Figure 16) are modified for climatic influence (Figure 17). Values for other frequencies can also be obtained (Table 12). Examples showing the application of the Cook's Method are presented in the next subsection.

Other Empirical Formulas--

In the past, engineers developed empirical formulas to determine design discharges for storm drains. Many such formulas took the general form:

$$Q = C\ A\ P\ \left(\frac{S}{A}\right)^x \text{ ---------------------------------------(14)}$$

where Q is the peak discharge in cfs, C is a coefficient, dependent on the physiographic and climatic conditions of the watershed, A is the drainage area in acres, P is the average rainfall

TABLE 10. RUNOFF CURVE NUMBER (CN), CONVERSIONS AND CONSTANTS[*]

CN For Condition II	CN for AMC I	CN for AMC II	I Values,[+] (in.)	Curve[+] Starts Where Wp = (in.)
100	100	100	0.000	0.00
98	94	99	0.204	0.04
96	89	99	0.417	0.08
94	85	98	0.638	0.13
92	81	97	0.870	0.17
90	78	96	1.11	0.22
88	75	95	1.36	0.27
86	72	94	1.63	0.33
84	68	93	1.90	0.38
82	66	92	2.20	0.44
80	63	91	2.50	0.50
78	60	90	2.82	0.56
76	58	89	3.16	0.63
74	55	88	3.51	0.70
72	53	86	3.89	0.78
70	51	85	4.28	0.86
68	48	84	4.70	0.94
66	46	82	5.15	1.03
64	44	81	5.62	1.12
62	42	79	6.13	1.23
60	40	78	6.67	1.33
58	38	76	7.24	1.45
56	36	75	7.86	1.57
54	34	73	8.52	1.70
52	32	71	9.23	1.85
50	31	70	10.0	2.00
48	29	68	10.8	2.16
46	27	66	11.7	2.34
44	25	64	12.7	2.54
42	24	62	13.8	2.76
40	22	60	15.0	3.00
38	21	58	16.3	3.26
36	19	56	17.8	3.56
34	18	54	19.4	3.88
32	16	52	21.2	4.24
30	15	50	23.3	4.66
25	12	43	30.0	6.00
20	9	37	40.0	8.00
15	6	30	56.7	11.34
10	4	22	90.0	18.00
5	2	13	190.0	38.00
0	0	0	Infinity	Infinity

[*] Reference: Soil Conservation Service (1972)

[+] For CN in column 1. I is the potential infiltration in inches and Wp is the rain fall in inches.

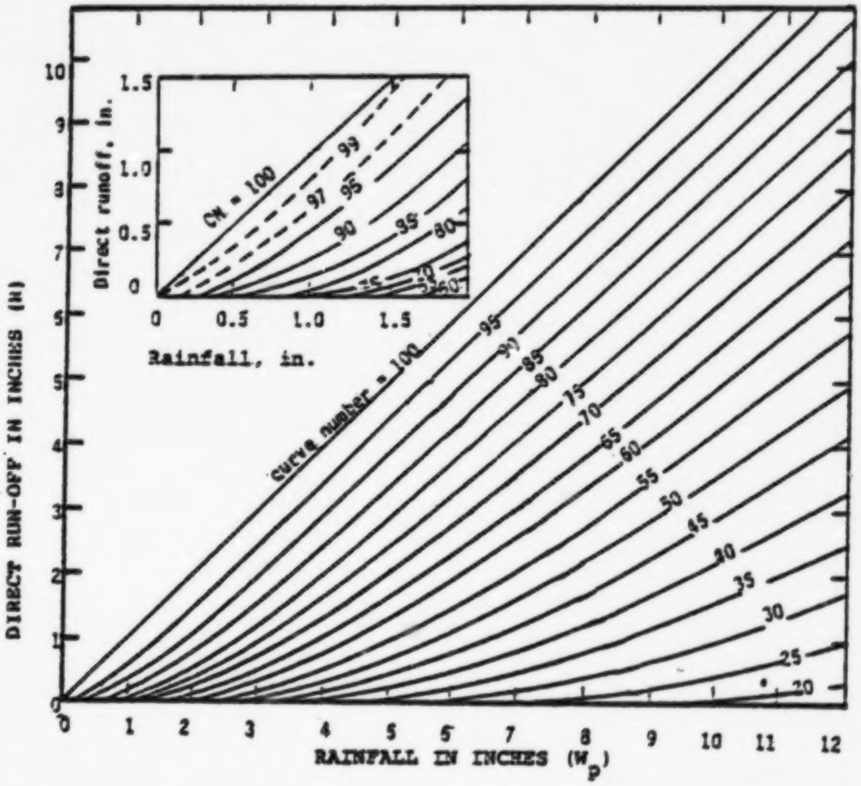

Figure 15. Estimation of direct run-off amounts from storm rainfall (Soil Conservation Service, 1972).

TABLE 11. INCREMENTAL W VALUES FOR USE IN COOK'S METHOD (Chow, 1964)

Watershed Characteristic	Extent or Degree	W
Relief	• Steep rugged terrain with average slopes generally above 30%	40
	• Hilly, with average slopes 10 to 30%	30
	• Rolling, with average slopes 5 to 10%	20
	• Relatively flat land, slopes 0 to 5%	10
Infiltration (I)	• No effective cover; either rock or thin soil mantle of negligible infiltration capacity	20
	• Slow to take up water; clay or other soil of low infiltration capacity	15
	• Deep loams with infiltration about that of typical prairie soils	10
	• Deep sand or other soil that takes up water readily and rapidly	5
Vegetal cover (C)	• No effective plant cover or equivalent	20
	• Poor to fair cover; clean cultivated crops or poor natural cover; less than 10% of the watershed in good cover	15
	• About 50% of watershed in good cover	10
	• About 90% of watershed in good cover, such as grass, woodlands, or equivalent	5
Surface storage	• Negligible; few surface depressions	20
	• Well-defined system of small drainage	15
	• Considerable depression storage with not more than 2% in lakes, swamps, or ponds	10
	• Surface-depression storage high; drainage system poorly defined; large number of lakes, swamps or ponds.	5

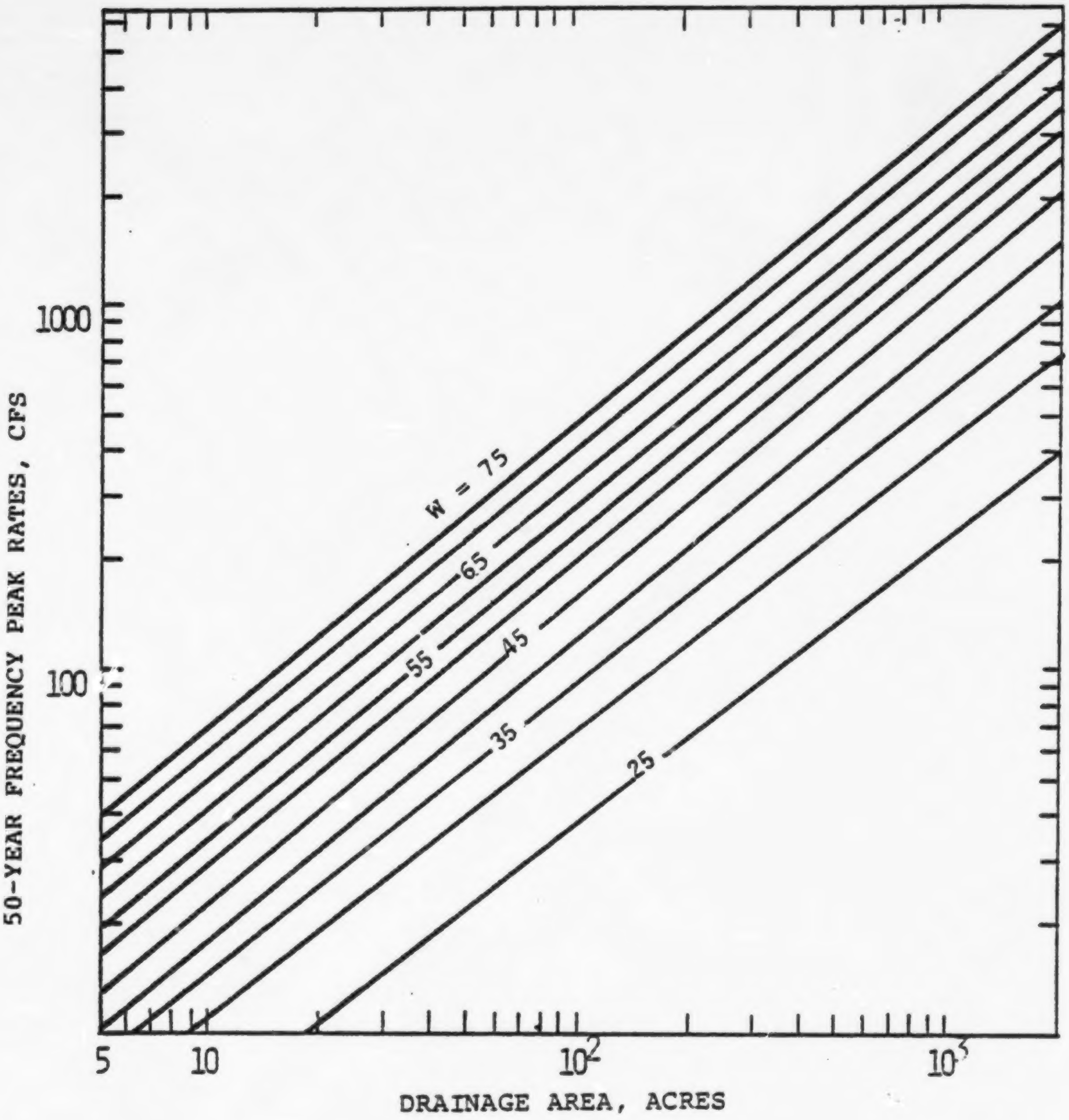

Figure 16. Chart for estimating 50-year-frequency peak rates of flow (Chow, 1964).

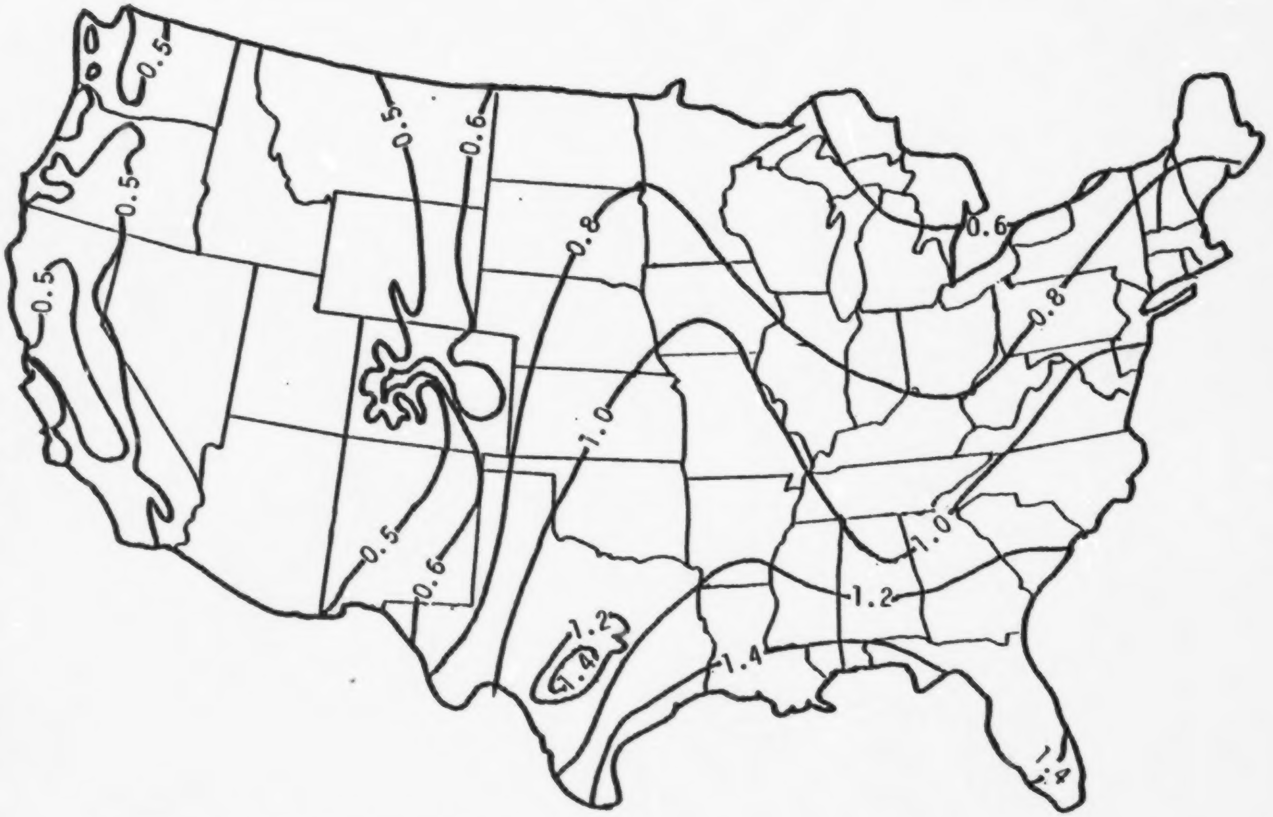
0.5
0.5
0.5
0.5
0.6
0.5
0.6
0.8
1.0
0.6
0.8
1.0
1.2
1.4
1.2
1.4
1.4

Figure 17. Distribution of rainfall factors used with the modified Cook's Method (Chow, 1964).

TABLE 12. FREQUENCY FACTORS FOR USE WITH COOK'S METHOD (CHOW 1964)

(I + C)*	Average annual precipitation, in.					
	10	20	30	40	60	80
	Ratio: 25-year/50-year					
5	0.31	0.38	0.41	0.44	0.48	0.51
10	0.41	0.50	0.55	0.58	0.63	0.66
15	0.50	0.59	0.64	0.68	0.73	0.77
20	0.55	0.65	0.71	0.76	0.82	0.87
25	0.60	0.71	0.78	0.83	0.90	0.92
30	0.64	0.76	0.83	0.89	0.92	0.92
35	0.67	0.81	0.89	0.92	0.92	0.92
40	0.71	0.85	0.92	0.92	0.92	0.92
	Ratio: 10-year/50-year					
5	0.05	0.08	0.10	0.12	0.15	0.17
10	0.10	0.16	0.21	0.24	0.30	0.34
15	0.16	0.25	0.31	0.37	0.45	0.51
20	0.21	0.33	0.42	0.49	0.60	0.68
25	0.26	0.41	0.52	0.61	0.75	0.80
30	0.31	0.49	0.62	0.74	0.80	0.80
35	0.36	0.58	0.73	0.80	0.80	0.80
40	0.42	0.66	0.80	0.80	0.80	0.80

* From Table 11.

intensity in in/hr, S is the slope of the drainage basin in ft per 1,000 ft, and X is an exponent. The following equations are two examples of the empirical formula (Chow, 1964).

Burkli-Ziegler Formula:

$$Q = C A P \left(\frac{S}{A} \right)^{0.25} \text{------------------------------} (15)$$

McMath Formula:

$$Q = C A P \left(\frac{S}{A} \right)^{0.5} \text{------------------------------} (16)$$

The C value of both formulas varies from 0.2 for previous rural areas to 0.75 for highly impervious metropolitan areas. Because C values for specific sites are not available, the use of these empirical formulas is not recommended.

Determining Infiltration

Numerous methods have been proposed for determining infiltration capacity and infiltration rate. Theoretical methods are not recommended because of their highly simplified nature and the requirement for determining special watershed properties. Three general approaches are available for the determination of landfill infiltration: actual measurement, estimation from surface runoff, and empirical calculation.

Infiltrometer Measurement--

Infiltrometers are preferred and are often used for determining infiltration rates of small watersheds, or experimental or sample areas within large watersheds. Two general types of infiltrometers may be used: (1) rainfall simulators, with the water applied in the form and at a rate comparable with natural rainfall, and (2) flooding infiltrometers, with the water applied in a thin sheet upon the enclosed area.

There are many modifications for rainfall simulators. The most common types in use are the F and FA types (Chow, 1964). They generally utilize plots of 1.8 m (6 ft) wide and 3.6 m (12 ft) long, respectively. Special nozzles are used to simulate the rainfall intensity and impact of raindrops. More detailed information can be found from hydrology literature.

The simplest type of flooding infiltrometer is a metal tube forced into the soil through a test section of the solid waste cover. Careful measurement of the amount of water added to hold a constant reservoir depth produces data for the infiltration curve. A more advanced type of flooding infiltrometer includes two concentric metal rings. Within the outer concentric ring, forms a second reservoir. This outside reservoir produces a buffer to minimize the lateral spreading of water below the inner tube. The tubes are usually 23 cm (9 in.) and

35 cm (14 in.) in diameter. Details of the flooding infiltrometers also can be found from hydrology literature.

Estimation From Surface Runoff--

Infiltration can be calculated as the difference between precipitation and surface runoff. Results for surface runoff, obtained by the methods described in the previous section, can be used in Equation (6) (in Table 2) to calculate the infiltration.

ASCE Method--

Most field infiltration approaches a steady-state minimum rate after one or two hours of precipitation. Relative minimum infiltration capacities for three broad soil groups (Table 13) are the basis of the ASCE empirical method (Jens, 1949).

TABLE 13. TYPICAL f_1 VALUES FOR BARE SOILS (Jens, 1949).

Soil Class	f_1* (in./hr)
High† (sandy soil)	0.50 - 1.00
Intermediate† (loam, clay, silt)	0.10 - 0.50
Low† (clay, clay loam)	0.10 - 0.10

* f_1 is the infiltration capacity after 1 hr of continuous rainfall (where 1 in./hr = 2.54 cm/hr).

† "High"	Besides sand, the open-structured soils of other textures, particularly the most friable silt loams are also included in this group.
"Intermetiate"	In this group, soils are associated with the loams. These loams contain considerable clay and much silt but are friable at ordinary water contents.
"Low"	Soils included are not only most clay and clay loams, but also other soils that are dense in structure. "Low" soils are commonly deeply cracked, and the initial infiltration rate may be high; however, soon after wetting, the cracks are closed and f_1 decreases rapidly to a more characteristic low rate.

Vegetal cover has considerable influence on the infiltration capacity of bare soil. Infiltration capacity can be increased 3 to 7.5 times by good permanent forest or grass cover, but little or no increase with poor row crops (Jens, 1949). A vegetation cover factor N (Table 14) is provided for the infiltration capacity of vegetated soil, f_{1V}

$$f_{1V} = N f_1 \text{ -- } (17)$$

where f_1 is the infiltration capacity after one hour of continuous rainfall.

Soil infiltration capacity is also affected by antecedent precipitation. However, this factor is not considered by this empirical method.

TABLE 14. VEGETATION COVER FACTOR FOR ESTIMATING INFILTRATION CAPACITY (Fenn, 1949)

Vegetation	Cover*	Cover Factor, N
Permanent forest and grass	Good	3.0 - 7.5
	Medium	2.0 - 3.0
	Poor	1.2 - 1.4
Close-growing crops	Good	2.5 - 3.0
	Medium	1.2 - 2.0
	Poor	1.1 - 1.3
Row crops	Good	1.3 - 1.5
	Medium	1.1 - 1.3
	Poor	1.0 - 1.1

* "Good"	Dense vegetal cover of high-quality grass having extensive root systems, properly managed in grass for several years.
"Medium"	30 - 80 percent of "Good" vegetal quality and density, well managed in grass for at least 2 years.
"Poor"	Less than 30 percent of "Good" vegetal density, low-quality grass and poor management.

Snowmelt Infiltration--

As discussed previously, it is generally assumed that surface runoff from snowmelt is rare in municipal landfills. In general, the water contained in snowfall on landfills either sublimates or melts to become landfill infiltrate. Two methods for estimating snowmelt are suggested: the degree day equations and the U.S. Army Corps of Engineers Equations.

The degree-day method provides a means of estimating the potential snowmelt on a daily basis. Local or transposed temperature and snowfall data are necessary. In its usual form, this equation is (Soil Conservation Service, 1972):

$$M = K\ (T - 32°) \qquad (18)$$

where M is the potential daily snowmelt in inches[*] of water equivalent, K is a constant representing the watershed condition (Table 15), and T is the average daily air temperature in °F.[†] The difference, T-32°, is the number of degree-days per day. For example, with an average daily temperature of 47°F, the number of degree-days for that day is 15. Because M is the potential snowmelt for the day, the actual snowmelt cannot be larger than the water equivalent available for the day. New snow may have a water equivalent of from 5 to 25 percent of snow depth. A value of 10 percent, or 1 in. of water for 10 in. of snow depth, is often used for making general estimates.

TABLE 15. VALUES OF THE CONSTANT K FOR DEGREE-DAY EQUATION (Soil Conservation Service, 1972)

Watershed Condition	K
Low runoff potential	0.02
Average heavily forested areas; north-facing slopes of open country	0.04 - 0.06
Average runoff potential	0.06
South-facing slopes of forested areas; average open country	0.06 - 0.08
High runoff potential	0.03

The U.S. Army Corps of Engineers has derived empirical equations similar to degree-day equation for the daily springtime snowmelt estimation (Chow, 1964). Either the mean daily temperature, T_{mean}, or the maximum daily temperature, T_{max}, is used, depending on the temperature ranges. Following equations are suggested:

---

[*] 1 in. = 2.54 cm.

[†] °F = (°C x 9/5) + 32.

For open sites:

$$M = 0.06\ (T_{mean} - 24) \text{------------------------------} (19)$$

$$M = 0.04\ (T_{max} - 27) \text{--------------------------------} (20)$$

For forested sites:

$$M = 0.05\ (T_{mean} - 32) \text{------------------------------} (21)$$

$$M = 0.04\ (T_{max} - 42) \text{------------------------------} (22)$$

For open areas (up to 10 percent of forest cover):

$$M = k'\ (0.00508\ I_i)\ (1-a) + (1-N)\ (0.0212\ T_a' - 9) + N(0.029\ T_c') + k\ (o.0084\ v)\ (0.22\ T_a' + 0.78\ T_d') \text{--} (23)$$

where:

M = snowmelt, in./day;
T_a' = difference between air temperature at 1-ft level and snow surface temperature, °F;
T_d' = difference between dew-point temperature at 10-ft level and snow surface temperature, °F;
v = wind speed at 50-ft level, mph;
I_i = observed or estimated solar radiation on horizontal surface, (langleys);
a = observed or estimated average snow surface albedo;
k' = basin short-wave radiation factor (between 0.9 and 1.1) depending on average exposure of open areas to short-wave radiation in comparison with an unshielded horizontal surface;
F = estimated average basin forest canopy cover expressed as a decimal fraction;
T_c' = difference between cloud base temperature and snow surface temperature, °F (estimated from upper-air temperature or by lapse rates from surface station, preferably on a snow-free site);
N = estimated cloud cover expressed as a decimal fraction
k = basin convection-condensation melt factor, depending on relative exposure of area to wind.

Determining Evapotranspiration

Thornthwaite (1955 and 1957) has suggested a term "potential evapotranspiration" to define the evapotranspiration that would occur when adequate soil moisture supply is available at all times. "Actual evapotranspiration" is the actual water loss, controlled by the amount of water actually available for plant use. There are many methods of estimating actual and potential evapotranspiration, but no one method can be applied generally for all purposes. All methods, however, fall into three general categories (1) theoretical approaches (based on physical processes), (2) analytical approaches (based on energy or water balance), and (3) empirical approaches (based on the site and climatic conditions). Methods which are applicable for the landfill conditions will be discussed.

Soil-Moisture Sampling--

Soil samples can be taken at different time intervals from different depths in the landfill cover for moisture content analysis.[*] After the moisture content of the soil samples is analyzed, the rate of change reflecting the water loss from evapotranspiration can be calculated. Various devices exist to measure soil moisture without the effort necessary for soil sampling.[†] These instruments have the further advantage of producing immediate results.

Lysimeter Measurements--

Evapotranspiration is estimated by growing the vegetation in lysimeters and then measuring the losses of water necessary to maintain satisfactory growth. Unfortunately, the lysimeter may not closely simulate the field conditions, and hence the results may not be reliable when applied to much larger areas. Modification of the lysimeter results for field application is thus necessary.

Pan Evaporation Adjusted--

The ratio of annual free water evaporation to annual pan evaporation, the "pan coefficient", averages nearly 0.7 (ranges from 0.67 to 0.81) (Linsley, et al., 1972). The rate of transpiration is approximately identical to the rate of evaporation from a free water surface provided the water availability for plants is not restricted (Chow, 1964; Linsley, et al., 1972). Estimated free water evaporation may, therefore, indicate the potential evapotranspiration from a vegetated soil surface. To obtain the annual distribution of potential evapotranspiration, the monthly pan evaporation is multiplied by the pan coefficient (usually 0.7). Obviously, estimations of evapotranspiration based on pan evaporation values are not sensitive to differences between vegetation types.

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[*] Standard procedure for moisture contents analysis: refer to "Annual Book of ASTM Standards", Part 19-Standard Method of Laboratory Determination of Moisture Contents of Soil D2216, pp. 338-339, 1980.

[†] Examples of instruments for in situ moisture content measurements are soil moisture tensiometer, soil mositure block (based on electrical resistance), and soil moisture cells. These instruments are available commercially.

Evapotranspiration Equations--

Many empirical or theoretical equations have been derived for estimating the evapotranspiration rates. Some typical equations suggested by Veihmeyer (1964) are listed in Table 16. These equations are usually based on the climate (temperature, humidity, etc.), available heat, and mean vegetation consumptive data.

Hedke equation--The Hedke equation (Equation (24) in Table 16) estimates evapotranspiration from the sum over the growing season of available heat. Available heat is the number of degree-days above the minimum growing temperature (usually 30 to 50°F). The use of this equation may be limited because of the lack of data for k and minimum growing temperatures.

Lowry-Johnson equation--This equation (equation (25) in Table 16) assumes a linear relationship between effective heat and evaporation. The effects of types and density of vegetation are not considered.

Blaney-Morin equation--This equation (equation (26) in Table 16) develops an empirical relation between evapotranspiration and percent day-time hours, mean monthly temperature and mean monthly relative humidity. The coefficient k (in equation (26)) varies with the type of vegetation (Table 17). The value of p can be computed from the U.S. Weather Bureau's "sunshine tables" (Table 18).

Blaney-Criddle equation--This equation (Equation (29) in Table 16) is similar to the Blaney-Morin equation, except that the former does not consider humidity. Tables 17 and 18 can also be used for equation (29) for the evapotranspiration calculation.

Thornthwaite equation--The Thornthwaite equation (equation (27) in Table 16) is based on an exponential relationship between mean monthly temperature and mean monthly consumption. This method is widely used by climatologists, geologists, and agriculture engineers. It has recently been adopted by sanitary engineers, because of its well developed information and procedures (Remson, et al., 1968; Fenn, et al., 1975; Dass, et al., 1977; and Lutton, et al., 1979). The relationship is based largely on experience in the central and eastern United States.

TABLE 16. EVAPOTRANSPIRATION EQUATIONS[*]

Name	Date	Period for E	Unit for E	Equation[†]	
Hedke Equation................	1930	Annual	Feet	$E = kH$	(24)
Lowry-Johnson Equation........	1942	Annual	Feet	$E = 0.000156H + 0.8$	(25)
Blaney-Morin Equation	1942	m months	Inches	$E = k \sum_{1}^{m} pt(114-h)$	(26)
Thornthwaite Equation	1944	Monthly	Centimeters	$E = 1.6 \left(\frac{10t}{TE}\right)^{a}$ where $a = 0.000000675(TE)^{3} - 0.0000771(TE)^{2} + 0.0179TE + 0.49239$	(27)
Penman Equation...............	1948	Daily	Millimeters	$E = \frac{AH - 0.27e}{A - 0.27}$ where $e = 0.35\,(e_a - e_d)(1 + 0.0098\omega_2)$ $H = R(1-r)(0.18 + 0.55S) - B(0.56 - 0.092e_d^{0.5})(0.10 + 0.90S)$	(28)
Blaney-Criddle Equation	1950	m months	Inches	$E = k\sum_{1}^{m} pt = kF$ where $F = \sum_{1}^{m} pt$	(29)

[*] Reference: after Veihmeyer, 1964.

[†] A = slope of saturated-vapor-pressure curve of air at absolute temperature in °F, or de_a/dt in mm Hg/°F.

B = a coefficient depending on temperature (see text for explanation).

e_a = saturation vapor pressure at mean air temperature in mm Hg.

e_d = saturation vapor pressure at mean dew point (i.e., actual vapor pressure in the air) in mm Hg, being equal to e_a multiplied by relative humidity in per cent.

e = daily evaporation in mm.

h = annual mean relative humidity in per cent, in Eq. (26).

H = accumulated degree-days above minimum growing temperature for growing season, in Eq.(24); or accumulated degree-days of maximum daily temperature above 32°F for growing season, in Eq. (27) or daily heat budget at surface in mm of water, in Eq. (28).

k = annual seasonal or monthly consumptive-use coefficient.

p = per cent of daytime hours of the year, occurring during the period, divided by 100 (Table 18).

r = estimated percentage of reflecting surface.

R = mean monthly extraterrestrial radiation in mm of water evaporated per day.

TE = Thornthwaite's temperature-efficiency index, being equal to the sum of 12 monthly values of heat index $i = (t/5)^{1.514}$, where t is mean monthly temperature in °C.

t = mean monthly temperature in °F, in Eqs. (26),(29) or in °C in Eq. (27).

E = evapotranspiration or consumptive use for given period.

ω_2 = mean wind velocity at 2 m above the ground in miles/day, or equal to $\omega_1(\log 6.6/\log h)$, where ω_1 is measured wind velocity in miles/day at height h in ft.

S = estimated ratio of actual duration of bright sunshine to maximum possible duration of bright sunshine.

Equation (27) gives only unadjusted rates of potential evaporation. Factors as listed in Table 19 can be applied to Equation (27) to obtain the adjusted potential evaporation.

Penman equation--This equation (Equation (28) in Table 16) is theoretically derived from the absorption of radiant energy by the ground surface. The values of e_a and A can be obtained from Figures 18 and 19, respectively. The values of R and B can be obtained from Tables 20 and 21, respectively.

TABLE 17. SEASONAL CONSUMPTIVE-USE COEFFICIENTS k IN BLANEY-CRIDDLE AND BLANEY-MORIN EQUATIONS, FOR IRRIGATED CROPS IN WESTERN UNITED STATES (Veihmeyer, 1964)

Crop	Length of growing season or period	k*
Alfalfa	Between frosts	0.80 - 0.85
Beans	3 months	0.60 - 0.70
Corn	4 months	0.75 - 0.85
Cotton	7 months	0.65 - 0.75
Flax	7-8 months	0.80
Grains, small	3 months	0.75 - 0.85
Sorghums	4-5 months	0.70
Orchard, citrus	7 months	0.50 - 0.65
Walnuts	Between frosts	0.70
Deciduous	Between frosts	0.60 - 0.70
Pasture, grass	Between frosts	0.75
Ladino clover	Between frosts	0.80 - 0.85
Potatoes	3½ months	0.65 - 0.75
Rice	3-5 months	1.00 - 1.20
Sugar beets	6 months	0.65 - 0.75
Tomatoes	4 months	0.70
Vegetables, small	3 months	0.60

* The lower values of k are for coastal areas; the higher values, for areas with arid climate.

Thornthwaite Tables--

Thornthwaite and Mather (1955; and 1957) developed several tables (see Appendix A) for calculation of evapotranspiration rates. The method is based on the assumptions described previously for the Thornthwaite equation (Equation (27)). Three steps are involved:

(1) A heat index is computed for each of the 12 months and summed to produce the annual index (Appendix A, Tables A-1);

TABLE 18. DAYTIME-HOURS PERCENTAGES, OR 100p, IN BLANEY-CRIDDLE AND BLANEY-MORIN EQUATIONS (Veihmeyer, 1964) (Annual value of p = 1.00)

Latitude, deg	J	F	M	A	M	J	J	A	S	O	N	D
North:												
60	4.67	5.65	8.08	9.65	11.74	12.39	12.31	10.70	8.57	6.98	5.04	4.22
50	5.98	6.30	8.24	9.24	10.68	10.91	10.99	10.00	8.46	7.45	6.10	5.65
40	6.76	6.72	8.33	8.95	10.02	10.08	10.22	9.54	8.39	7.75	6.72	6.52
35	7.05	6.88	8.35	8.83	9.76	9.77	9.93	9.39	8.36	7.87	6.97	6.86
30	7.30	7.03	8.38	8.72	9.53	9.49	9.67	9.22	8.33	7.99	7.19	7.15
25	7.53	7.14	8.39	8.61	9.33	9.23	9.45	9.09	8.32	8.09	7.40	7.42
20	7.74	7.25	8.41	8.52	9.15	9.00	9.25	8.96	8.30	8.18	7.58	7.66
15	7.94	7.36	8.43	8.44	8.98	8.80	9.05	8.83	8.28	8.26	7.57	7.88
10	8.13	7.47	8.45	8.37	8.81	8.60	8.86	8.71	8.25	8.34	7.91	8.10
0	8.50	7.66	8.49	8.21	8.50	8.22	8.50	8.49	8.21	8.50	8.22	8.50
South:												
10	8.86	7.87	8.53	8.09	8.18	7.86	8.14	8.27	8.17	8.62	8.53	8.88
20	9.24	8.00	8.57	7.94	7.85	7.43	7.76	8.03	8.13	8.76	8.87	9.33
30	9.70	8.33	8.62	7.73	7.45	6.96	7.31	7.76	8.07	8.97	9.24	9.86
40	10.27	8.63	8.67	7.49	6.97	6.37	6.76	7.41	8.02	9.21	9.71	10.49

TABLE 19. MEAN POSSIBLE DURATION EXPRESSED IN UNITS OF 30 DAYS OF 12 HR EACH, OR THE ADJUSTING FACTOR FOR POTENTIAL EVAPOTRANSPIRATION COMPUTED BY THE THORNTHWAITE EQUATION (Veihmeyer, 1964).

Latitude, deg	J	F	M	A	M	J	J	A	S	O	N	D
0	1.04	0.94	1.04	1.01	1.04	1.01	1.04	1.04	1.01	1.04	1.01	1.04
10	1.00	0.91	1.03	1.03	1.08	1.06	1.08	1.07	1.02	1.02	0.98	0.99
20	0.95	0.90	1.03	1.05	1.13	1.11	1.14	1.11	1.02	1.00	0.93	0.94
30	0.90	0.87	1.03	1.08	1.18	1.17	1.20	1.14	1.03	0.98	0.89	0.88
35	0.87	0.85	1.03	1.09	1.21	1.21	1.23	1.16	1.03	0.97	0.86	0.85
40	0.84	0.83	1.03	1.11	1.24	1.25	1.27	1.18	1.04	0.96	0.83	0.81
45	0.80	0.81	1.02	1.13	1.28	1.29	1.31	1.21	1.04	0.94	0.79	0.75
50	0.74	0.78	1.02	1.15	1.33	1.36	1.37	1.25	1.06	0.92	0.76	0.70

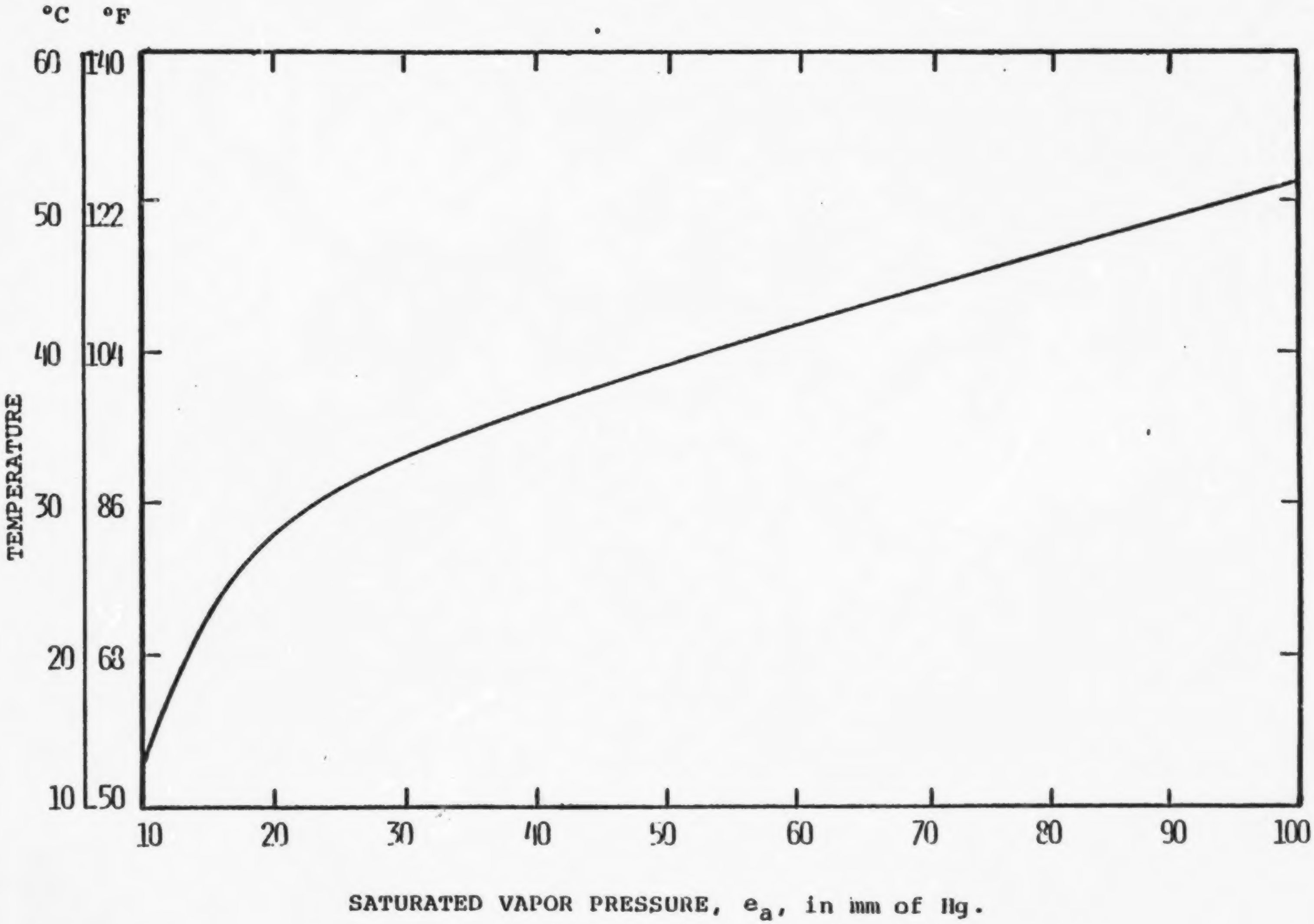


Figure 18. Temperature versus saturated vapor pressure (Veihmeyer, 1964)

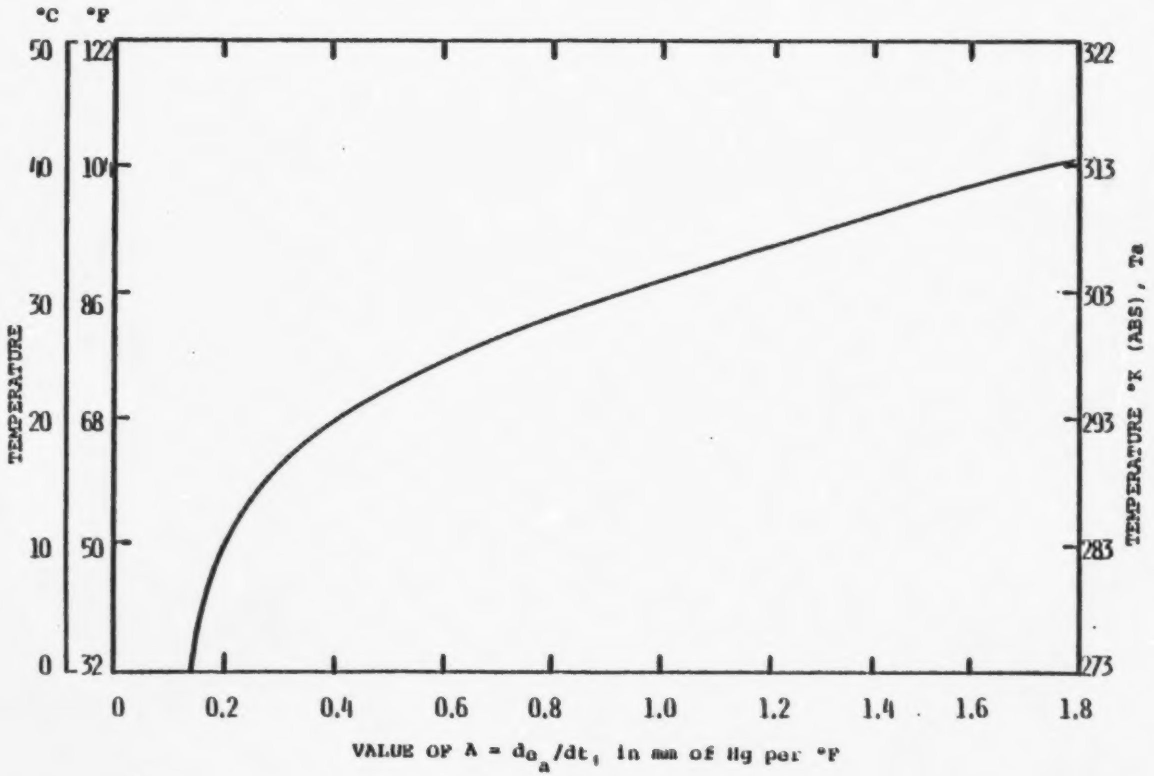


Figure 19. Temperature versus A for use in the Penman equation (Equation (28), Veihmeyer, 1964).

TABLE 20. MIDMONTHLY INTENSITY OF SOLAR RADIATION ON A HORIZONTAL SURFACE, IN MILLIMETERS OF WATER EVAPORATED PER DAY, OR THE VALUE R IN THE PENMAN EQUATION (Veihmeyer, 1964)

Latitude, deg	J	F	M	A	M	J	J	A	S	O	N	D
North												
90	--	--	--	7.9	14.9	18.1	16.8	11.2	2.6	--	--	--
80	--	--	1.8	7.8	14.6	17.8	16.5	10.6	4.0	0.2	--	--
70	--	1.1	4.3	9.1	13.6	17.0	15.8	11.4	6.8	2.4	0.1	--
60	1.3	3.5	6.8	11.1	14.6	16.5	15.7	12.7	8.5	4.7	1.9	0.9
50	3.6	5.9	9.1	12.7	15.4	16.7	16.1	13.9	10.5	7.1	4.3	3.0
40	6.0	8.3	11.0	13.9	15.9	16.7	16.3	14.8	12.2	9.3	6.7	5.5
30	8.5	10.5	12.7	14.8	16.0	16.5	16.2	15.3	13.5	11.3	9.1	7.9
20	10.8	12.3	13.9	15.2	15.7	15.8	15.7	15.3	14.4	12.9	11.2	10.3
10	12.8	13.9	14.8	15.2	15.0	14.8	14.8	15.0	14.9	14.1	13.1	12.4
0	14.5	15.0	15.2	14.7	13.9	13.4	13.5	14.2	14.9	15.0	14.6	14.3
South												
10	15.8	15.7	15.1	13.8	12.4	11.6	11.9	13.0	14.4	15.3	15.7	15.8
20	16.8	16.0	14.6	12.5	10.7	9.6	10.0	11.5	13.5	15.3	16.4	16.9
30	17.3	15.8	13.6	10.8	8.7	7.4	7.8	9.6	12.1	14.8	16.7	17.6
40	17.3	15.2	12.2	8.8	6.4	5.1	3.3	7.5	10.5	13.8	16.5	17.8
50	17.1	14.1	10.5	6.6	4.1	2.8	1.2	5.2	8.5	12.5	16.0	17.8
60	16.6	12.7	8.4	4.3	1.9	0.8	--	2.9	6.2	10.7	15.2	17.5
70	16.5	11.2	6.1	1.9	0.1	--	--	0.8	3.8	8.8	14.5	18.1
80	17.3	10.5	3.6	--	--	--	--	--	1.3	7.1	15.0	18.9
90	17.6	10.7	1.9	--	--	--	--	--	--	7.0	15.3	19.3

TABLE 21. VALUES OF B IN THE PENMAN EQUATION (Veihmeyer, 1964)

Ta. °abs.	B, mm water/day	Ta, °F	B, mm[*] water/day
270	10.73	35	11.48
275	11.51	40	11.96
280	12.40	45	12.45
285	13.20	50	12.94
290	14.26	55	13.45
295	15.30	60	13.96
300	16.34	65	14.52
305	17.46	70	15.10
310	18.60	75	15.65
315	19.85	80	16.25
320	21.15	85	16.85
325	22.50	95	18.10
		100	18.80

[*] $B = \sigma Ta^4$, where σ is Boltzmann constant, or 2.01×10^{-9} mm/day, and Ta is the air temperature. Heat of evaporation was assumed to be constant at 500 cal/g of water.

(2) Daily unadjusted potential evapotranspiration is obtained from the heat index and the tables (Appendix A, Tables A-2 and A-3);

(3) Finally, the potential evapotranspiration is adjusted for month and day lengths with correction factors provided in table (Appendix A, Table A-4).

As indicated before, this method has been widely used by sanitary engineers. However, the accuracy of this method for sanitary landfills is less studied. Examples of this method and the accuracy of the results will be more thoroughly discussed later in this report.

Determining Water Storage Capacity

Water storage capacity can be expressed as:

Storage Capacity = Field Capacity - Wilting point-----(30)

In agricultural situations, the root zone is up to the order of 4 ft thick. At a refuse disposal site, the root zone thickness will be limited by the landfill cover due to the inhospitality of the refuse to vegetation root systems (Lutton, *et al.*, 1979). Therefore, Equation (30) is applied only to the landfill cover. For a refuse cell, the storage capacity is equal to the field capacity.

The Water storage capacity (or, the "available water" for vegetation) of the landfill cover depends on the soil type, soil compaction, and cover thickness. Figure 20 summarizes the three states of soil water through the range of USDA soil types. According to this figure, 2-ft silty loam cover will store about 4 in. of water:

Storage Capacity = 2 ft (3.4 in./ft field capacity - 1.4 in./ft wilting point)
= 4 in.

For a refuse cell, the refuse composition, size, and density are major factors affecting field capacity (Stone, 1974; and Fungaroli, 1971). Figure 21 is a plot of refuse field capacity versus refuse dry density for both processed (milled) and unprocessed refuse. Refuse field capacity increases as refuse dry density increases. When refuse grain size decreases, field capacity also increases significantly. Results indicate that, for unprocessed refuse, if the dry density of refuse is around 300 $\#/yd^3$, the field capacity would be roughly 2.5 in. of water per foot of refuse depth.

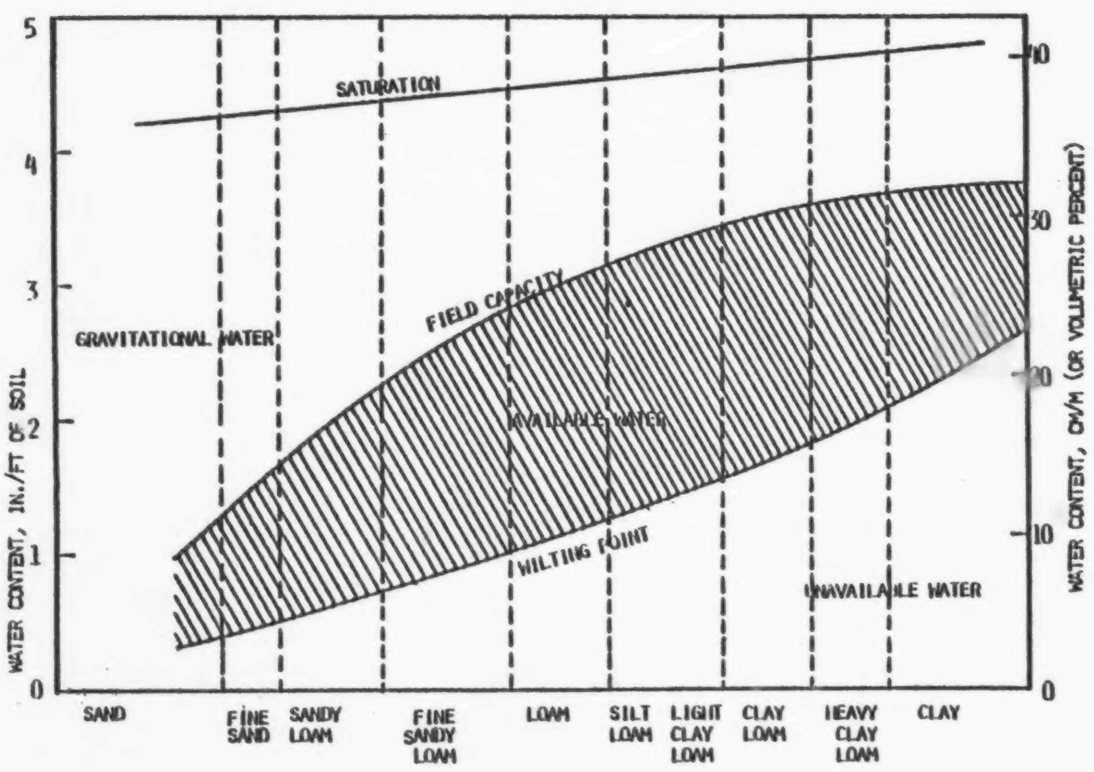


Figure 20. Water storage capacity of USDA soils (Lutton, et al., 1979).

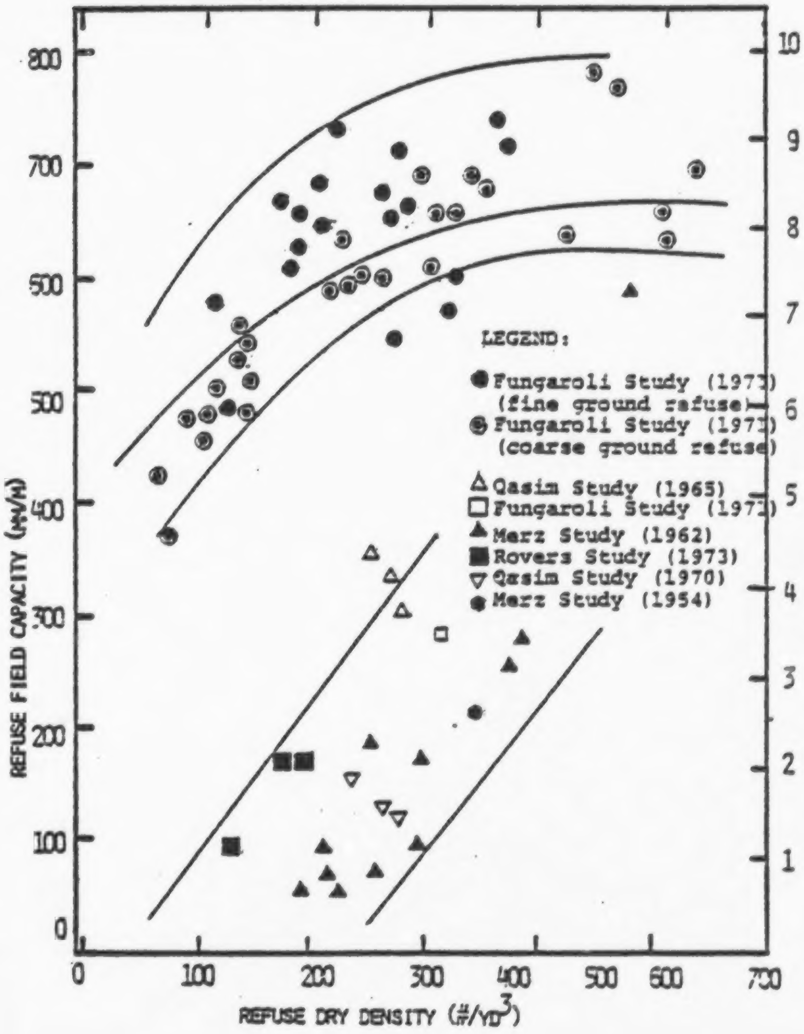


Figure 21. Refuse field capacity versus refuse dry density.

The above data, however, illustrates only the apparent field capacity of the refuse, i.e., the amount of water absorbed beyond the original refuse moisture content. The absolute refuse field capacity (or dry weight field capacity) is higher than the apparent value. Fenn, et al. (1975) suggested that the absolute field capacity of unprocessed municipal refuse varies from about 2.4 in./ft to about 4.2 in./ft (or 20 to 35 percent* of the moisture content by volume). Several analyses performed on municipal refuse found its original moisture content ranging from 10 to 20 percent by volume (Fenn, et al., 1975). Thus, the apparent field capacity of municipal refuse might range from 0 to 3 in./ft† of refuse.

Stone (1974) has conducted numerous water absorption studies for various refuse components. The initial moisture content and total moisture holding capacity are reported and summarized in Table 22. Results are presented in units of weight of water per weight of refuse. Stone (1974) indicated that the field capacity of any refuse can be estimated with reasonable accuracy if the distribution of their components is known. Results of Table 22 can be used for this estimation. An example of the field capacity estimation for refuse from the City of Oceanside, California, is presented in Table 23.

Determining Time of First Appearance of Leachate

If no channeling effect occurred in the refuse, leachate will first appear when the soil cover and refuse reach field capacity. The time required for the first appearance of leachate can be obtained by using the moisture-routing calculation (details of the calculation procedures will be described in the next sub-section).

The time of first appearance of leachate can also be estimated using graphics prepared by Fungaroli (1971). The graphical procedures are summarized as follows:

1. Moisture routing through the soil cover
   a. Determine type of cover soil;
   b. Determine the thickness;
   c. Using Figure 22, evaluate the time (t_1) from placement when leachate will pass through the soil cover if the soil is placed at the wilting moisture content;

---

* 2.4 in./ft = 2.4 in./12 in. = 20%
4.2 in./ft = 4.2 in./12 in. = 35%

† 20% absolute moisture content - 20% original moisture content = 0% apparent moisture content = 0 in./ft.
35% absolute moisture content - 10% original moisture content = 25% apparent moisture content = 3 in./ft.

TABLE 22. WATER ABSORPTION RANGES FOR SOLID WASTE COMPONENTS (Stone, 1974)

Component	Moisture content, percent dry weight					
	Water absorption capability			Total moisture-holding capacity[*]		
	Maximum	Average	Minimum	Maximum	Average	Minimum
Newsprint[†]		290			290[‡]	
Cardboard (solid and corrugated)[†]		170			170[‡]	
Other miscellaneous paper	400		100	400[‡]		100[‡]
Lawn clippings (grass and leaves)	200		60	370		140
Shrubbery, tree prunings	100		10	250		10
Food waste (kitchen garbage)	100		0	300		0
Textiles (cloth of all types, rope)	300		100	300[‡]		100[‡]
Wood, plastic, glass metal (all inorganic)		0			0	

[*] Calculated from water absorption plus initial moisture content in as-received samples.

[†] Sample variation was negligible.

[‡] Initial moisture contents as-received were less than 6 percent in the laboratory tests; therefore, they were considered negligible compared to the variation in moisture absorbed.

TABLE 23. PREDICTED RANGE OF ABSORPTIVE CAPACITY OF MUNICIPAL REFUSE AS RECEIVED AT OCEANSIDE, LANDFILL (Stone, 1974)

Component	Total moisture-holding capacity as determined in laboratory tests[*]		Available field absorption capacity of waste components[†]		Average composition[‡] (percent)	Field absorptive capacity[#]	
	Maximum	Minimum	Maximum	Minimum		Maximum	Minimum
Newsprint	290		262		7.2	19	
Cardboard	170		146		8.3	12	
Miscellaneous paper	400	100	397	97	23.6	94	23
Leaves and grass	370	140	312	92	3.8	12	4
Prunings	250	10	207	0	6.3	13	0
Garbage (food waste)	300	0	229	0	9.2	21	0
Textiles	300	100	284	84	2.3	7	2
Non-absorbents[**]	0	0	0	0	39.3	0	0
Total					100.0	178	60

[*] Oven-dried samples, from Table 22, percent dry wt basis.

[†] The absorptive capacities determined in laboratory tests reduced by the measured moisture contents from Oceanside waste samples, percent dry wt basis.

[‡] Average of year's (four quarters) composition of collected refuse arriving at Oceanside Municipal Landfill Site.

[#] Pounds water per 100 pounds of average mixed refuse as received at the landfill; derived from product of available absorptive capacity and average composition for each component.

[**] Includes wood (absorption very slow), foam plastic (insignificant quantity), and dirt, sand, and ashes (which entrain but do not absorb).

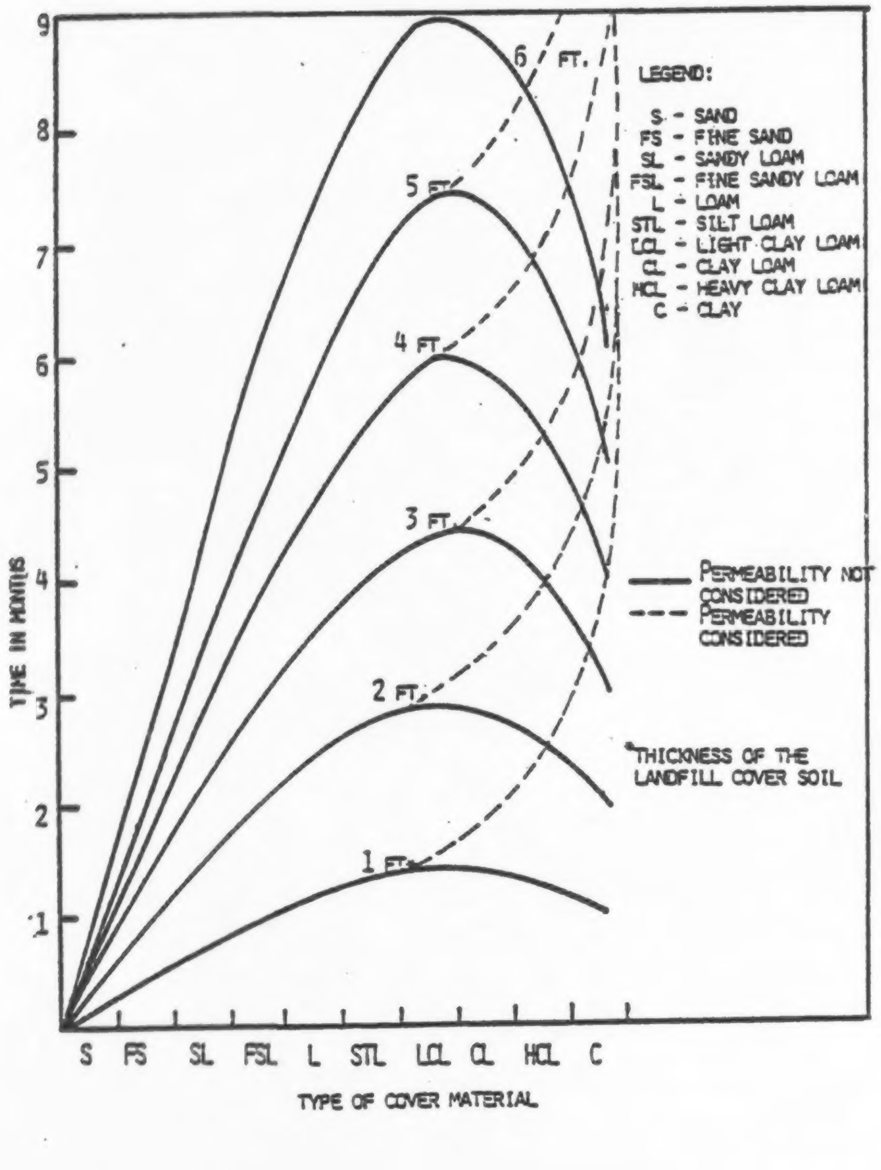


Figure 22. Time needed for one inch infiltration per month to bring soil from wilting to field capacity (Fungaroli, 1971).

d. Calculate the time (t_1) to reach field capacity, if the placement moisture is higher than the wilting moisture:

$$t_1 = t \times \frac{\Delta S_p}{\Delta S_w}$$

where t = calculated time from Figure 22;

ΔS_p = field capacity - placement moisture content;

ΔS_w = field capacity - wilting moisture content.

2. Moisture routing through refuse:

   a. Determine field capacity of the refuse;
   b. Determine the original moisture content of the refuse;
   c. Calculate apparent field capacity of the refuse (field capacity-original moisture content);
   d. Determine depth of refuse;
   e. Using Figure 23 determine the time (t_2) from placement when leachate will appear at the bottom of the refuse.

3. Total time for leachate appearance:

   a. Add times from computations 1 and 2 (i.e., $t_1 + t_2$) to get time required for one inch infiltration per month to bring the soil and refuse to field capacity;
   b. Determine the monthly average infiltration for a given area;
   c. Total time in months = ($t_1 + t_2$) x monthly average infiltration.

Determining Ground-Water Intrustion

Determination of ground-water intrusion into a landfill (i.e., underflow) can be obtained by a hydrologic investigation and calculations using Darcy's Law. Because the rate and direction of the underflow through the landfill may be different from those in surrounding ground water, ground-water monitoring wells should be drilled on and adjacent to a landfill site. Figure 24 illustrates a typical arrangement of monitoring wells. Wells A and D are off-site wells used for determining local ground water characteristics. These wells can also be used to determine soil permeability. Wells B and C serve a similar purpose for the actual landfill. For monitoring studies, measurement of soil and refuse permeabilities are the most critical factors for underflow estimations. Permeability may be obtained by pumping tests,

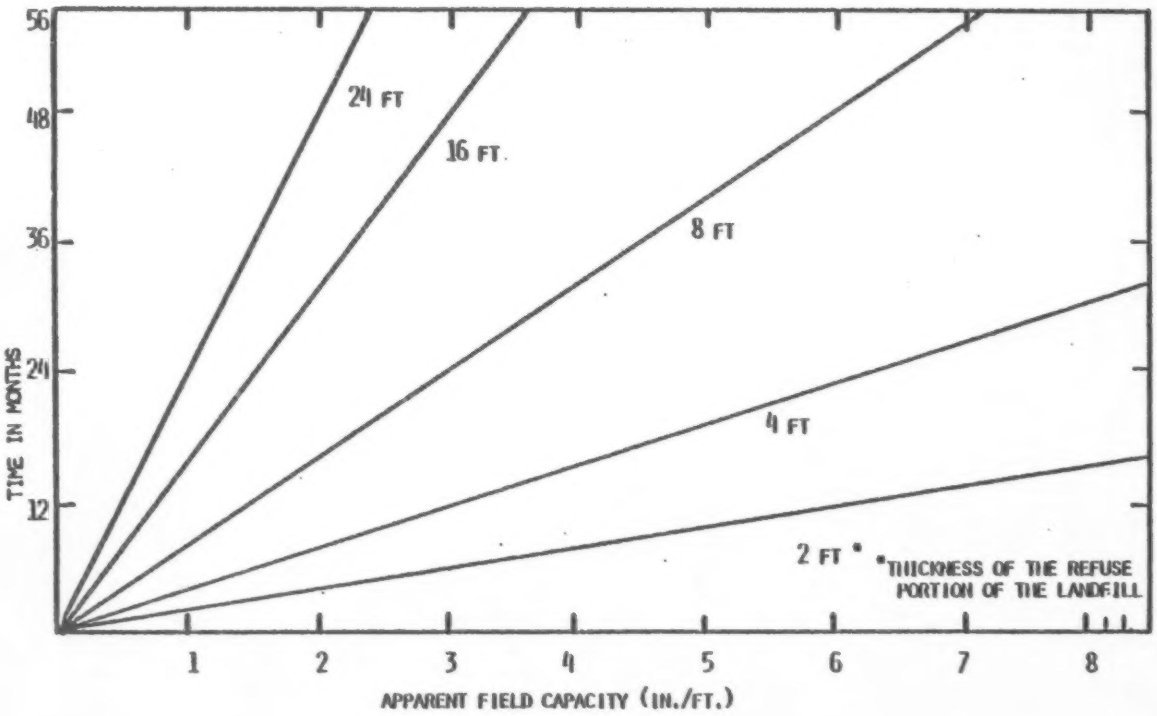
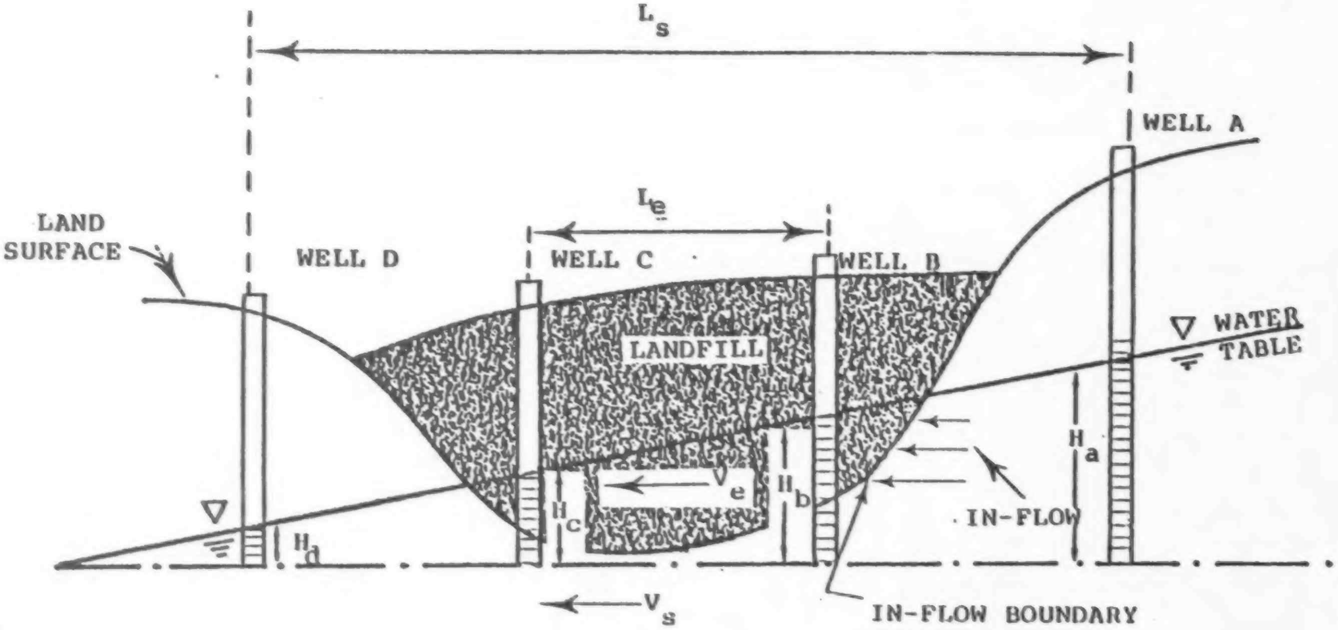


Figure 23. Time needed for percolate to pass through the refuse (Fungaroli, 1971).


L_s
WELL A
L_e
LAND SURFACE
WELL D
WELL C
WELL B
LANDFILL
WATER TABLE
H_a
V_e
H_b
H_c
H_d
IN-FLOW
IN-FLOW BOUNDARY
V_s

Legend:

V_e = underflow velocity;

V_s = local ground-water flow velocity;

$\frac{H_b - H_c}{L_e}$ = on-site hydraulic gradient;

$\frac{H_a - H_d}{L_s}$ = off-site hydraulic gradient;

A = projection area of the in-flow boundary on a plane perpendicular to the in-flow direction.

Figure 24. Estimating rate of landfill underflow (see text for explanation).

injection tests, tracer studies, or by testing of individual soil samples removed during drilling (Chow, 1964).

After ground-water characteristics (most notably ground-water flow pattern and depth to ground water) and soil permeabilities have been determined, Darcy's Law can be applied for identifying velocity and rate of the underflow:

$$V = K \frac{dh}{dL} \text{ ------------------------------- (1)}$$

and $$Q = KA \frac{dh}{dL} \text{ ----------------------------- (2)}$$

where V and Q are the volocity and rate of underflow, respectively, K is the permeability and dh/dL is the hydraulic gradient (Figure 24). For estimation of the leachate generation rate by underflow, the landfill underflow projection area (A), perpendicular to the ground-water flow direction, must also be approximated. This area is calculated using the projection of the in-flow (to the landfill) boundary area onto a plan perpendicular to the in-flow direction.

EXAMPLES OF LEACHATE GENERATION MODELS

Leachate generation calculations are presented in this subsection. The calculation (Figure 25) involves two major steps: (1) quantification of leachate generation factors (precipitation, surface runoff, infiltration, etc.), and (2) water balance calculations.

Quantification of Leachate Generation Factors

Some of the methods (Figure 25, under "Alternative Methods") for quantification of leachate generation factors have been explained in the previous sub-section. In the following pages only those methods requiring clarification will be illustrated by examples.

Curve Number Method--

Example 1--A small landfill having silty-clay soil (permeability = 10^{-5} cm/sec) and poor pasture cover had a rainfall of 2.5 in. The 5-day total antecedent rainfall was 0.4 in. Estimate the direct runoff for this storm.

Solution:

1. Using Table 7, find the antecdent moisture condition = AMC I.
2. Using Table 8, determine the hydrologic soil group = D.
3. Using Table 9, identify CN = 89 for AMC = II.
4. Using Table 10, convert CN for AMC = I, that is CN = 77.

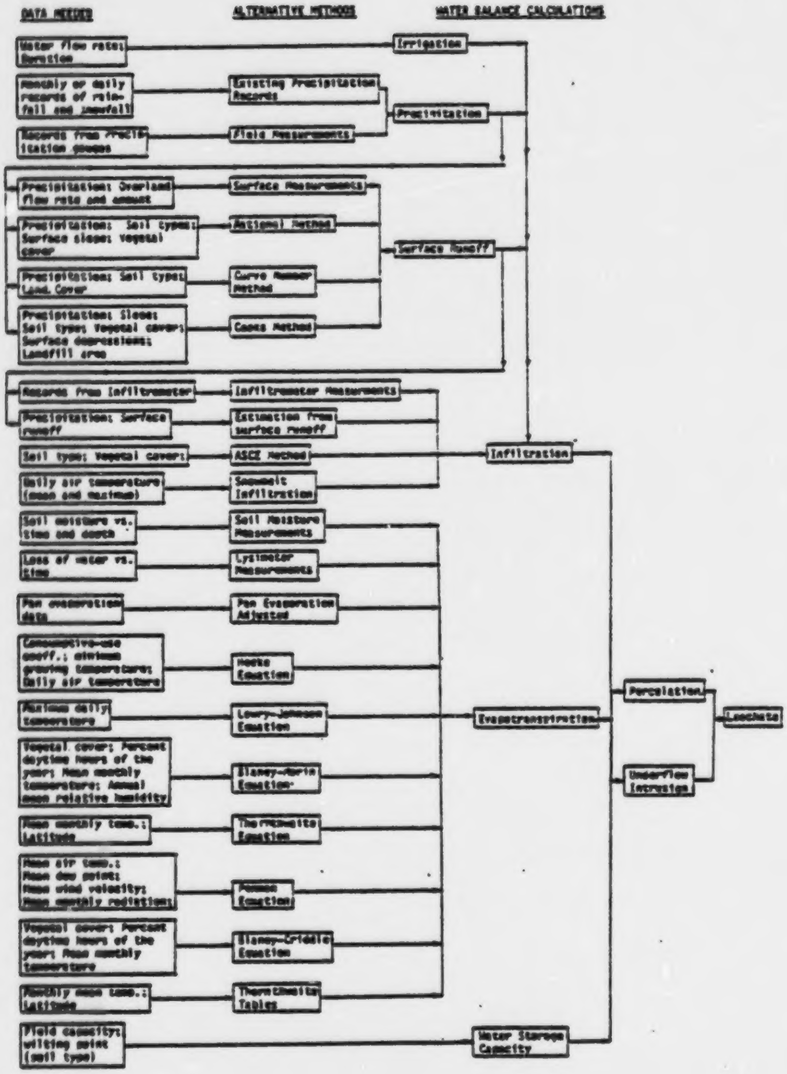

DATA NEEDED
ALTERNATIVE METHODS
WATER BALANCE CALCULATIONS
Water flow rate; Duration
Irrigation
Monthly or daily records of rainfall and snowfall
Existing Precipitation Records
Precipitation
Records from Precipitation gauges
Field Measurements
Precipitation; Overland flow rate and amount
Surface Measurements
Precipitation; Soil types; Surface slope; Vegetal cover
Rational Method
Surface Runoff
Precipitation; Soil types; Land Cover
Curve Number Method
Precipitation; Slope; Soil type; Vegetal cover; Surface depressions; Landfill area
Cooks Method
Records from Infiltrometer
Infiltrometer Measurements
Precipitation; Surface runoff
Estimation from surface runoff
Soil type; Vegetal cover;
ASCE Method
Infiltration
Daily air temperature (mean and maximum)
Snowmelt Infiltration
Soil moisture vs. time and depth
Soil Moisture Measurements
Loss of water vs. time
Lysimeter Measurements
Pan evaporation data
Pan Evaporation Adjusted
Consumptive-use coeff.; minimum growing temperature; Daily air temperature
Hedke Equation
Maximum daily temperature
Lowry-Johnson Equation
Evapotranspiration
Percolation
Leachate
Underflow Intrusion
Vegetal cover; Percent daytime hours of the year; Mean monthly temperature; Annual mean relative humidity
Blaney-Morin Equation
Mean monthly temp.; Latitude
Thornthwaite Equation
Mean air temp.; Mean dew point; Mean wind velocity; Mean monthly radiation;
Penman Equation
Vegetal cover; Percent daytime hours of the year; Mean monthly temperature
Blaney-Criddle Equation
Monthly mean temp.; Latitude
Thornthwaite Tables
Field capacity; wilting point (soil type)
Water Storage Capacity

Figure 25. Flow chart of water balance calculations.

5. Entering Figure 13 or Equation (13) with W_p = 2.5 in. and CN = 77, find R = 0.74 in.

When a storm occurs over several days, the runoff estimate may be calculated on a daily basis and changes the AMC accordingly. The total runoff can also be estimted directly by using the value of CN applicable for the first day of the storm and the total storm rainfall for the calculation period. Such an estimate will differ slightly from that made with a day-by-day calculation.

Cooks Method--

Example 2--A 40-acre landfill located in central Ohio has an average surface slope of 5 percent. The landfill surface is covered with a silty-clay soil with no vegetation. The average annual precipitation in the landfill area is 36 in. Determine the peak rate of flow with a predicted 10-year frequency.

Solution:

1. Determine the coefficient W. Using Table 11, the component values of W are:

   Relief----------------------------10
   Infiltration----------------------15
   Cover-----------------------------20
   Surface storage-------------------20
   W =65

2. Entering Figure 16 with 40 acres at curve W = 65, read a discharge of 150 cfs for a 50-year frequency rate.

3. Determine the rainfall factor from Figure 17 for this location. For central Ohio, this is 0.75.

4. Determine the frequency factor from Table 12. Enter the Table with the sum of W for the given conditions of infiltration and cover soil (I + C = 15 + 20 = 35) and the annual precipitation of 36 in. The 10-year frequency ratio is 0.77.

5. Compute the 1-year-frequency peak flow = 150 cfs x 0.75 x 0.77 = 87 cfs.

ASCE Infiltration Method--

Example 3--A small landfill has the conditions described in Example 1. Find the total infiltration capacity of a 2.5 in. storm lasting four hours.

Solution:

1. Using Table 13, estimate f_1 = 0.3 in./hr.
2. Using Table 14, estimate N = 1.3.
3. Calculate $f_{1v} = N \cdot f_1$ = 0.39 in./hr.
4. Total infiltration capacity = 0.39 in./hr x 4 hr. = 1.56 in.

Degree-day method--

Example 4--A snow storm leaves 20 in. of snow on a municipal landfill with a low runoff potential. Find the potential snowmelt infiltration during the ten days after the storm. The average daily air temperatures in °F for these ten consecutive days are: 30, 28, 34, 34, 36, 37, 40, 42, 44, and 45.

Solution:

1. Using Table 15, find K = 0.02.
2. Converting snowfall to water equivalent = 20 in. x 1 in./10 in. = 2 in.
3. Using Equation (18) find the total potential snowmelt infiltration = 0.02 [(34-32) + (34-32) + (36-32) + (37-32) + (40-32) + (42-32) + (44-32) + (45-32)] = 1.12 in.
4. Since 1.12 in. is smaller than 2 in. (maximum water equavalent as calculated in Step 2 above), so 1.12 in. is the answer.

Pan evaporation adjusted--

Example 5--The monthly pan evaporation rates in a municipal landfill area are shown in the following table. Find the potential evapotranspiration from the landfill.

Solution:

Multiply pan evaporation rate by 0.7

Month	Pan Evaporation	Potential Evapotranspiration
January	0.2 in.	0.14 in.
February	0.2 in.	0.14 in.
March	0.6 in.	0.42 in.
April	1.5 in.	1.05 in.
May	2.7 in.	1.89 in.
June	4.2 in.	2.94 in.
July	4.8 in.	3.36 in.
August	4.5 in.	3.15 in.

Month	Pan Evaporation	Potential Evapotranspiration
September	3.6 in.	2.52 in.
October	2.1 in.	1.47 in.
November	0.9 in.	0.63 in.
December	0.1 in.	0.07 in.
Annual Total	25.4 in.	17.78 in.

Thornthwaite Tables--

Example 6--A municipal landfill located in Chester County, Pennsylvania (latitude 40°N) has a mean monthly temperature as illustrated in Table 24 (row 1). Find the monthly potential evapotranspiration.

Solution:

1. Using Appendix A, Table A-1, find the monthly heat index (row 2 of Table 24).
2. Using Appendix A, Table A-2, find the unadjusted potential evapotranspiration (row 3 of Table 24).
3. Using Appendix A, Table A-4, find the correction factor for monthly duration of sunlight (row 4 of Table 24).
4. Monthly potential evaporation (row 5) = daily unadjusted potential evapotranspiration (row 3) x correction factor (row 4).

Water Balance Calculation

The water balance calculation can be made at several levels of detail according to the needs of the designer or regulatory agencies. Two types of calculation can be employed: gross estimation, and moisture routing methods.

Gross Estimation--

The gross estimation method assumes that the moisture contents of the entire landfill (including refuse and cover soils) are equal or exceed field capacity during the considered period. Leachate generation can be grossly predicted by using the water balance equation (Table 2). The period of consideration can be a year, a month, or some specially selected period (e.g., the duration of a storm). The following equation calculates the gross estimation of the annual leachate generation (Ann. L):

$$\text{Ann. L} = \text{Ann. } W_p - \text{Ann. R} - \text{Ann. E} \quad \text{---------------------(31)}$$

where:

Ann.L = annual leachate generation;

Ann.W_p = annual precipitation;

Ann.R = annual surface runoff; and

TABLE 24. POTENTIAL EVAPOTRANSPIRATION COMPUTED BY THORNTHWAITE METHOD

Item	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.
Mean monthly Temperature (°C)	0.8	1.0	5.2	11.2	17.1	21.9	24.4	23.5	19.8	13.7	7.4	1.7
Monthly heat index	0.06	0.09	1.06	3.39	6.44	9.36	11.02	10.41	8.03	4.60	1.81	0.20
Daily unadjusted potential evapotranspiration (mm)	0	0.1	0.5	1.4	2.5	3.5	4.0	3.8	3.0	1.8	0.8	0.1
Correction factor (Table A-4)	25.2	24.9	30.9	33.3	37.2	37.5	38.1	35.4	31.3	28.8	24.9	24.3
Monthly potential evapotranspiration (mm)	0	2.5	15.5	46.6	93.0	131.3	152.4	134.5	93.6	51.8	19.9	2.4

Ann. E = annual evapotranspiration.

Example 7--Assume that the 20-year average annual precipitation (Ann. W_p) at Chippewa Falls, Wisconsin is 24.5 in. The runoff coefficient for this site is estimated as 0.07 and the annual evaporation is 28 in. Find the annual leachate generation.

Solution:

1. Ann. R = 24.5 in. x 0.07 = 1.7 in. (rational method).
2. Ann. E = 28 in. x 0.7 = 19.6 in. (pan evaporation adjusted method).
3. Using Equation (31),
   Ann. L = 24.5 in. - 1.7 in. - 19.6 in. = 3.2 in.

Moisture routing method--

The moisture routing method can be used for landfills which exceed field capacity and for those which do not. The method begins with calculations for moisture routing through the soil cover and uses the results to calculate moisture routing through the underlying compacted refuse. The following example shows the procedures for a hypothetical landfill which does not exceed field capacity (Remson, et al., 1968).

Example 8--A hypothetical landfill is composed of 8 ft (2.4 m) of compacted refuse and 2 ft (0.6 m) of silt loam cover. The field capacity and wilting point for the cover soil are 34.9 and 10 percent respectively (volume basis). The inital moisture content and field capacity for the refuse are 3.9 and 28.6 percent respectively (volume basis). The landfill and cover are completed on July 1 with an initial soil moisture content of 10 percent (wilting point). Find the leachate movement within one year.

Solution:

1. Moisture routing through soil cover--Table 25 shows the routing of moisture through the landfill cover for the first year according to the Thornthwaite Method. Computation is made month by month and results are summarized in Table 25 and explained as follows:
   a. Find the mean monthly temperature (T) and precipitation data (W_p) for the study period (row 1 and 2 of Table 25).
   b. Find the potential evapotranspiration (E) (row 3, calculation method refer to Example 6, Table 24).

TABLE 25. MOISTURE ROUTING THROUGH A SANITARY LANDFILL SOIL COVER

ITEM	PARAMETER	JULY	AUG.	SEPT.	OCT.	NOV.	DEC.	JAN.	FEB.	MAR.	APR.	MAY	JUNE	ANNUAL
1	Mean monthly temperature, T (°C)	24.4	23.5	19.8	13.7	7.4	1.7	0.8	1.0	5.2	11.2	17.1	21.9	
2	Monthly precipitation, P (mm of water)	108	142	100	74	90	77	86	75	102	85	90	103	1132
3	Potential evapotranspiration, E (mm)	152.4	134.5	93.6	51.8	19.9	2.4	0	2.5	15.5	46.6	93.0	131.3	743.5
4	Runoff, R (mm)	16	21	15	11	14	12	13	11	15	13	14	15	170
5	Moisture available for infiltration, I (mm)	92	121	85	63	76	65	73	64	87	72	76	88	962
6	I - E (mm)	-60.4	-13.5	-8.6	11.2	56.1	62.6	73	61.5	71.5	25.4	-17.0	-43.3	218.5
7	NE = Σ neg (I - E) (mm)	-60.4	-73.9	-82.5								-17.0	-60.3	
8	Soil moisture storage, S_s (mm)	60	60	60	71.2	127.3	150	150	150	150	150	133	99.7	39.7
9	Change in storage, ΔS_s (mm)	0	0	0	11.2	56.1	22.7	0	0	0	0	-17	-33.3	
10	Actual evapotranspiration, AE (mm)	92	121	85	51.8	19,9	2.4	0	2.5	15.5	46.6	93.0	121.30	651
11	PER_s (mm)	0	0	0	0	0	39.9	73	61.5	71.5	25.4	0	0	271.3

c. Find Runoff (R): For this example the runoff is calculated by the Rational Method, assuming a runoff coefficeint of 0.15 (using Table 3, under heavy soil, and 2% slope).

d. Find the moisture available for infiltration (I): $I = W_p - R$ (row 5 of Table 25).

e. Find the value of I - E (row 6).

f. Find the accumulated potential water loss (PE): $PE = \Sigma$ negative (I - E) (row 7)

g. Find the soil moisture storage (S_s): The initial value is equal to the initial moisture content of the cover soil (10% x 0.6 m = 60 mm). For humid areas (defined as areas where the sum of all the I - E values is positive, the initial value is equal to field capacity of landfills older than one year (see Example 10). This initial value of S_s is assigned to the last month having a positive value of I - E. In dry areas (see Example 11), the initial and subsequent S_s values must be determined from the appropriate table (Table A-8 to A-16 of Appendix A) using the value of PE calculated per row 7 above.

   Other S_s values in row 8 of Table 25 can be found from appropriate tables (Table A-8 to A-16) if I - E is negative. (If I - E is positive, representing additions of moisture to the soil, it must be added to the previous months S_s value.) No S_s value can exceed the soil field capacity (in this example, 150 mm).

h. Change in soil moisture storage (ΔS_s): represents the change in soil moisture from month to month (row 9).

i. Actual evapotranspiration (AE): when $I - E > 0$, $AE = E$; when $I - E < 0$, $AE = E + (I - E) - \Delta S_s$. AE soil moisture is depleted, the actual evapotranspiration rate (AE) decreases below its potential rate (E).

j. Percolation in cover soil (PER_s): when $S_s <$ field capacity, $PER_s = 0$; when $S_s >$ field capacity, $PER_s = (I - E) - \Delta S_s$.

2. Moisture routing through compacted refuse Table 26 shows the routing of moisture through the refuse

TABLE 26. MOISTURE ROUTING THROUGH UNDERLYING COMPACTED REFUSE (FIRST YEAR)

Depth Below Landfill Cover (m)	End-of-month moisture content (mm)											
	July	Aug.	Sept.	Oct.	Nov.	Dec.	Jan.	Feb.	Mar.	Apr.	May	June
0 - 0.3	12[*]	12	12	12	12	51.9[‡]	86[*]	86	86	86	86	86
0.3 - 0.6	12	12	12	12	12	12	50.9[‡]	86	86	86	86	86
0.6 - 0.9	12	12	12	12	12	12	12	38.4	86	86	86	86[†]
1.2 - 1.5	12	12	12	12	12	12	12	12	35.9	61.3	61.3	61.3
1.5 - 1.8	12	12	12	12	12	12	12	12	12	12	12	12
1.8 - 2.1	12	12	12	12	12	12	12	12	12	12	12	12
2.1 - 2.4	12	12	12	12	12	12	12	12	12	12	12	12
Leachate Production	0	0	0	0	0	0	0	0	0	0	0	0

[*] 0.3 m x 3.9% = 12 mm (initial moisture content per layer).
0.3 m x 28.6% = 86 mm (field capacity per layer).

[†] location of wetting front (i.e., depth at which the field capacity is exceeded).

[‡] 12 mm + 39.9 mm (from Table 25) = 51.9 mm.
51.9 mm + 73 mm (from Table 25) - 86 mm + 12 mm = 50.9 mm.

for the first year after placement. The landfill is divided into eight 0.3-m layers for computational purposes and has an initial moisture content of 12 mm/per layer. The field capacity for each layer is 86 mm (0.3 m x 28.6% = 86 mm). Thus, 74 mm of moisture are required to bring each layer of refuse from its initial content to field capacity. Table 26 describes the moisture content in each layer at the end of the month. The moisture surpluses are routed from the soil cover (Table 25) into the uppermost layer. When a layer has reached field capacity, the excess moisture is routed into the next layer below. The results shown in Table 26 indicate the wetting front is at a depth of approximately 0.9 m one year after placement.

Example 9--Determine the time required for the first appearance of leachate for the landfill described in Example 8 by the moisture routing method.

Solution:

Moisture routing through the soil cover after the first year of emplacement is calculated in Table 27. Calculation methods are largely identical to those described in Example 8. Different results are obtained for rows 7 to 11 because of the different PE data for July, August, and September.

Table 28 shows the moisture routing through the underlying refuse for the second year. Two years after the emplacement, the wetting front is at a depth of 2.1 m.

Table 29 illustrates moisture routing through the underlying refuse for the third year. Leachate is produced in January (Table 29), that is, leachate generation occurs 31 months after refuse placement.

Example 10--A landfill located in Cincinnati, Ohio has a 0.6m clay-loam cover with 250 mm/m of available water. Runoff coefficients for this landfill are 0.17 for the wet season and 0.13 for the dry season. Assuming that the field capacity of the underlying refuse has been reached, find the yearly leachate production.

Solution:

The calculation procedures for this example are similar to Example 8. This landfill is located in a humid area (Table 30, Σ (I – E) >0). The initial value of potential water loss (PE) is zero because the landfill is at field capacity at the end of the rainy season. Negative values of I – E are then accumulated. Because the refuse is at field capacity, percolation through the

TABLE 27. MOISTURE ROUTING THROUGH A LANDFILL SOIL COVER AFTER ONE YEAR OF EMPLACEMENT

Row	Parameter	JULY	AUG.	SEPT.	OCT.	NOV.	DEC.	JAN.	FEB.	MAR.	APR.	MAY	JUNE
1 to 6	(Same as that listed in Table 25)												
7	PE*	-120.7*	-134.2*	0	0	0	0	0	0	0	0	-17	-60.3
8	S_s	65.3	60	57	68.2	124.3	150	150	150	150	150	133	99.7
9	ΔS_s	-34.4	-5.30	-3.0	11.2	56.1	25.7	0	0	0	0	-17	-33.3
10	AE	126.4	126.3	88.0	51.8	19.9	2.4	0	2.5	15.5	46.6	93.0	121.3
11	PER_s	0	0	0	0	0	36.9	73	61.5	71.5	25.4	0	0

* PE = Σ negative (I—E) (refer to Example 8)

$- 60.3 \text{ mm} + (-60.4 \text{ mm}) = -120.7 \text{ mm}$

$- 120.7 \text{ mm} + (-13.5 \text{ mm}) = -134.2 \text{ mm}$

TABLE 28. MOISTURE ROUTING THROUGH UNDERLYING COMPACTED REFUSE (SECOND YEAR)*

Depth Below Landfill Cover (m)	End-of-month moisture content (mm)											
	July	Aug.	Sept	Oct.	Nov.	Dec.	Jan.	Feb.	Mar.	Apr.	May	June
0 - 0.3	86	86	86	86	86	86	86	86	86	86	86	86
0.3 - 0.6	86	86	86	86	86	86	86	86	86	86	86	86
0.6 - 0.9	86	86	86	86	86	86	86	86	86	86	86	86
0.9 - 1.2	61.3	61.3	61.3	61.3	61.3	86	86	86	86	86	86	86
1.2 - 1.5	12	12	12	12	12	24.2	86	86	86	86	86	86
1.5 - 1.8	12	12	12	12	12	12	23.2	84.7	86	86	86	86
1.8 - 2.1	12	12	12	12	12	12	12	12	82.2	86	86	86
2.1 - 2.4	12	12	12	12	12	12	12	12	12	33.6	33.6	33.6
Leachate Production	0	0	0	0	0	0	0	0	0	0	0	0

---

* Refer to Table 26 for first year moisture routing data.

TABLE 29. MOISTURE ROUTING THROUGH UNDERLYING COMPACTED REFUSE (THIRD YEAR)

Depth Below Landfill Cover (m)	July	Aug.	Sept.	Oct.	Nov.	Dec.	Jan.	Feb.	Mar.	Apr.	May	June
0 - 0.3	86	86	86	86	86	86	86	86	86	86	86	86
0.3 - 0.6	86	86	86	86	86	86	86	86	86	86	86	86
0.6 - 0.9	86	86	86	86	86	86	86	86	86	86	86	86
0.9 - 1.2	86	86	86	86	86	86	86	86	86	86	86	86
1.2 - 1.5	86	86	86	86	86	86	86	86	86	86	86	86
1.5 - 1.8	86	86	86	86	86	86	86	86	86	86	86	86
1.8 - 2.1	86	86	86	86	86	86	86	86	86	86	86	86
2.1 - 2.4	33.6	33.6	33.6	33.6	33.6	70.5	86	86	86	86	86	86
Leachate Production*	0	0	0	0	0	0	57.5	61.5	71.5	25.4	0	0

* Total leachate produced in the third year = 57.5 mm + 61.5 mm + 71.5 mm + 25.4 mm
= 215.9 mm

TABLE 30. WATER BALANCE CALCULATION FOR A LANDFILL IN CINCINNATI, OHIO (Fenn, et al., 1975)

Parameter*	JAN.	FEB.	MAR.	APR.	MAY	JUNE	JULY	AUG.	SEPT.	OCT.	NOV.	DEC.	ANNUAL
E	0	2	17	50	102	134	155	138	97	51	17	3	766
W_p	80	76	89	82	100	106	97	90	73	65	83	84	1025
Runoff Coeff.	0.17	0.17	0.17	0.17	0.17	0.13	0.13	0.13	0.13	0.13	0.13	0.17	
R	14	13	15	14	17	14	13	12	9	8	11	14	154
I	66	63	75	68	83	92	84	78	64	57	72	70	872
I - E	+66	+61	+58	+18	-19	-42	-71	-60	-33	+6	+55	+67	+106
PE				(0)	-19	-61	-132	-192	-225				
S_s	150	150	150	150	131	99	61	41	33	39	94	150	
ΔS_s	0	0	0	0	-19	-32	-38	-20	-8	+6	+55	+56	
AE	0	2	17	50	102	124	122	98	72	51	17	3	658
L†	+66	+61	+57	+18	0	0	0	0	0	0	0	+11	213

* All units in mm (See Table 25 for explanation)
† L = leachate production

soil cover will eventually become leachate. Leachate might be formed in the month of January through April and December for the given landfill conditions (Table 30). The annual leachate production will be 213 mm.

Example 11--A landfill located in Los Angeles, California has a 0.6 m silty loam and moderately deep-rooted grass cover. The runoff coefficient is 0.15 for only those months were W_p >E and surface runoff is negligible for the dry months. Find the potential leachate generation.

Solution:

The available water for cover soil (Figure 20) is 200 mm/m. For landfills in arid areas ($\Sigma(I - E)<0$), the soil moisture at the end of the wet season is below field capacity. Therefore, it is necessary to find an initial value of PE for accumulating negative values of I - E. Thornthwaite's method of successive approximations is used (Thornthwaite, et al., 1957). The S_s values (Table 31) for negative PE are found from Table A-10 of Appendix A (soil moisture storage = 200 mm/m x 0.6 m = 120 mm at field capacity).

HSSWDS Model

Due to tedious procedures involved for the calculation of leachate generation factors and moisture routing data, attempts have been made in the past to develop computer simulation models for the estimation of leachate generation. Lutton et al. (1979) 1979) adopted the USDAHL Model from USDA Hydrograph Laboratory (HL) and applied this model to actual landfill conditions. A more complete model, which was designed especially for landfill sites, is the HSSWDS (Hydrologic Simulation on Solid Waste Disposal Sites) model, developed by Perrier, et al. (1980). The HSSWDS Model was modified from the SCS curve number runoff method and the hydrologic portion of the USDA-SEA hydrologic model (CREAMS) (Knisel, 1980). The model is intended to assist the owners or operators of solid waste sites, or permit officials, in arriving at a logical, well-documented decision.

The flowchart for the HSSWDS Model is shown in Figure 26. From minimal input data, the model will simulate daily, monthly, and annual values of runoff, infiltration, evapotranspiration, and soil water storage. The model stores many default values of parameters estimates to be used when measured and existing data are not available; for example, soil-water characteristics, precipitation, mean monthly temperatures, mean monthly solar radiation, and vegetative characteristics. Five years of climatic records for a large number of stations within the United States are on tape for easy access to be used in lieu of onsite measurements.

TABLE 31. WATER BALANCE CALCULATION FOR A LANDFILL IN LOS ANGELES, CA (Fenn, *et al*., 1975)

Parameter*	J	F	M	A	M	J	J	A	S	O	N	D	Annual
E	34	36	49	59	76	94	117	115	96	73	52	39	840
W_p	78	79	66	27	9	2	0	1	5	14	29	68	378
Runoff Coeff.	0.15	0.15	0.15	0	0	0	0	0	0	0	0	0.15	
R	12	12	10	0	0	0	0	0	0	0	0	10	44
I	66	67	56	27	9	2	0	1	5	14	29	58	334
I-E	+32	+31	+7	-32	-67	-92	-117	-114	-91	-59	-23	+19	-506
PE			>-39	-71	-138	-230	-347	-461	-552	-661	-634		
S_s	52	83	90	70	40	19	7	3	1	1	1	20	
ΔS_B	+32	+31	+7	-20	-30	-21	-12	-4	-2	0	0	+19	
AE	34	36	49	47	39	23	12	5	7	14	29	39	334
L†	0	0	0	0	0	0	0	0	0	0	0	0	0

* All units in mm (see Table 25 for explanation)

† L = leachate production

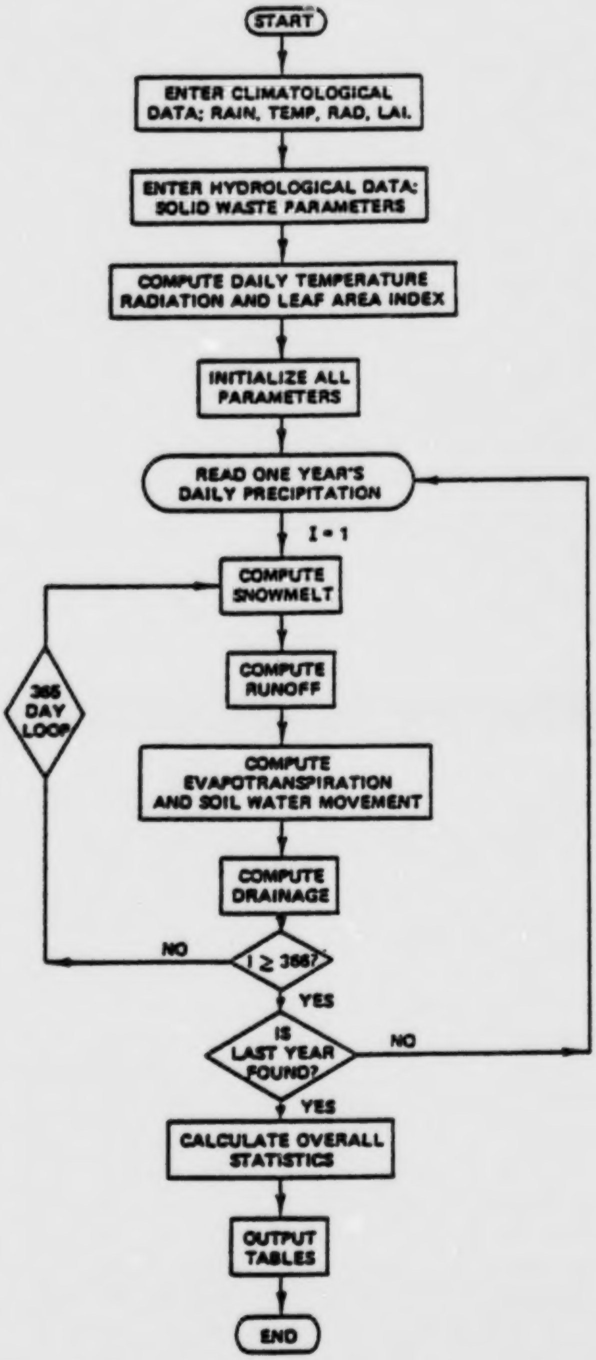

START
ENTER CLIMATOLOGICAL DATA; RAIN, TEMP, RAD, LAI.
ENTER HYDROLOGICAL DATA; SOLID WASTE PARAMETERS
COMPUTE DAILY TEMPERATURE RADIATION AND LEAF AREA INDEX
INITIALIZE ALL PARAMETERS
READ ONE YEAR'S DAILY PRECIPITATION
I = 1
COMPUTE SNOWMELT
365 DAY LOOP
COMPUTE RUNOFF
COMPUTE EVAPOTRANSPIRATION AND SOIL WATER MOVEMENT
COMPUTE DRAINAGE
NO
I ≥ 366?
YES
IS LAST YEAR FOUND?
NO
YES
CALCULATE OVERALL STATISTICS
OUTPUT TABLES
END

Figure 26. Generalized flowchart for the hydrologic simulation Model HSSWDS (Perrier, et al., 1980).

Like other leachate quantification methods, the HSSWDS Model also applied the water balance principle. The techniques or formula used by this model are as follows:

- Precipitation--Using data from existing precipitation records or actual field measurements;
- Surface runoff--Using the SCS curve number technique;
- Snow-melt estimation--Using the following formula:

  $$M_i = 0.18T \text{------------------------} (32)$$

  where M_i is the snowmelt on day i; and T is the temperature (°F) above freezing. This relation is used unless M is greater than the amount of surface snow;
- Evapotranspiration--Using the Penman equation (refer to Table 16); and
- Infiltration and percolation--Using the soil moisture routing technique for the cover soil. The cover soil is divided into seven layers for routing. Each soil layer is subject to evapotranspiration. The equation used for calculation is shown below:

  $$DR_i = FR_i + SM_{i-1} - SM_i - ET_i + M_i \text{------} (33)$$

  where DR_i = percolation (or drainage) in soil on day i;
  FR_i = infiltration on day i;
  SM_i = soil water storage on day i;
  ET_i = evapotranspiration on day i; and
  M_i = amount of snowmelt on day i.

  (Note that Equation (33) is similar to Equation (7) as discussed previously). The moisture content of the solid waste material is assumed at field capacity; therefore, the volume of water entering the solid waste by percolation through the cover soil will immediately generate leachate.

The user of the model can request a final cover soil with a vegetative and a barrier layer or with a uniform final cover soil. The user can select an impermeable liner separating the final cover soil material from the solid waste cells and select the life expectancy of the liner. No prior experience with computer programming is required for model usage. The reader is referred to the original report (Perrier, et al.,

1980) for details of the application of the model.

DISCUSSION

Although the water balance method has been widely used for estimating landfill leachate, the accuracy and sensitivity of the method for the actual landfill are less studied. More than one hundred different approaches are available for water balance calculation (refer to Figure 25); however, no comparisons have been made in the past to identify which approach can achieve better results, or which approach is suitable for what types of landfill conditions. The applicability, accuracy, and sensitivity of the water balance method are discussed in the following text.

Sensitivi[illegible] Analysis

In conducting water balance calculations, assumed or subjective estimations of leachate generation factors or conditions (e.g., runoff coefficient or curve number for runoff estimation; f_1 or N factors for ASCE infiltration estimation; consumptive-use coefficient for evapotranspiration estimation; etc.) are usually involved. Changes of certain assumptions or estimations may have little effect on the final leachate quantities calculated; however, sometimes these changes may lead to great differences in the calculated results. Table 32 shows examples of such effects (Dass, *et al.*, 1977). Cases 1 and 2 of Table 32 shows that an increase of the runoff coefficient from 0.065 to 0.200 would reduce the calculated annual leachate data from 3.75 to 2.04 inches, respectively, for Blue Valley Sanitary Landfill. Cases 1 and 3 indicate that an assumed annual evapotranspiration of 15 in. (from bare soil in the Blue Valley Sanitary Landfill area) instead of 20 in. (from grain crops in the same area) could increase the calculated leachate quantity from 3.75 to 7.95 in. (about 112% difference). The same table also shows that for Cases 1 and 4, the changes of available moisture of cover soil from 1.05 to 4.80 in. (more than four times difference) only affects the leachate estimation about 20 percent (from 3.75 to 3.01 in.).

Perrier, *et al.* (1980) also conducted a sensitivity analysis for the HSSWDS Model for a landfill in Cincinnati, Ohio. Overall effects of varying linear life, curve number, winter cover factor, depth of barrier and vegetative soils, leaf area index, barrier soil compaction, and soil texture, on leachate generation and other leachate generation factors, were studied. Summarization of this study is given in Table 33.

It should be noted that the sensitivity of the water balance calculation is site specific. Even two sites with similar refuse and soil conditions, radically different climatological and hydrological data may change the sensitivity results greatly.

TABLE 32. LEACHATE PRODUCTION AT BLUE SANITARY LANDFILL UNDER DIFFERENT ASSUMPTIONS, IN INCHES OF WATER* (Dass, *et al*., 1977)

Case	Runoff Coefficient	Annual Potential Evapotranspiration	Maximum Soil Moisture Available For Plant Use	Annual Leachate
1	0.065	20.0	1.05	3.75
2	0.200	20.0	1.05	2.04
3	0.065	15.0	1.05	7.94
4	0.065	20.0	4.80	3.01

* Methods for quantification of leachate generation factors:
Surface runoff--rational method;
Evapotranspiration--Thornthwaite Tables; and
Infiltration--estimation from surface runoff.
(1 in. - 25.4 mm.)

TABLE 33. SUMMARY OF SENSITIVITY STUDY RESULTS (Perrier, *et al*., 1980)

Parameter	Change		Surface Runoff			Evapotranspiration			Leachate Generation			Soil Percolation			Type of Variable
	From	To	Sensi-tivity	Direc-tion	Rank*	Sensi-tivity	Direc-tion	Rank	Sensi-tivity	Direc-tion	Rank	Sensi-tivity	Direc-tion	Rank	
Impermeable liner	5 yr	Ind.#	NA**	NA	NA	NA	NA	NA	†	↑	1	†	↓	1	Computed
SCS curve number	81	99	††	↑	1	††	↓	2	‡	↓	3	NA	NA	NA	Constant
Winter cover factor	0.5	1.0	‡	↓	2	‡‡	↑	1	†	↓	3	NA	NA	NA	Seasonal
Depth of barrier soil	6 in.	18 in.	‡‡	↑	1	‡	↓	2	†	↓	3	NA	NA	NA	Constant
Depth of vegetative soil	12 in.	36 in.	†	V** ↓	3	§	↑	2	§	↓	1	NA	NA	NA	Constant
Leaf area index	Excell	Brgd#	§	↑	2	‡‡	↓	1	‡	↑	3	NA	NA	NA	Seasonal
Barrier soil Compaction	NCP#	CP#	‡	↑	2	†	↓	3	§	↓	1	NA	NA	NA	Constant
Soil Texture#															
Vegetative layer-S	S	C	†	V↓	1	†	V↓	2	†	V↑	3	NA	NA	NA	Constant
Vegetative layer-SL	L	C	†	↓	1	†	↓	2	†	V↓	3	NA	NA	NA	
Vegetative layer-L	SLC	C	†	↓	1	†	↓	3	†	↓	2	NA	NA	NA	
Vegetative layer-SLC	NCP	CPD	§	↑	1	†	↓	3	§	↓	1	NA	NA	NA	

Note: Arrow indicates direction of changes, (↑ increase and ↓ decrease).

[*] Rank means the percentage change when the parameter is related to the average annual precipitation (1 = largest, 3 = lowest change).

[**] NA = Not affected, and V = Variable (Arrow indicates general tendancy)

† = Slightly

‡ = Moderately

§ = Significantly

‡‡ = Highly

†† = Extremely

[#] Ind. = indefinite life; Brgd = bare ground; NCP = not compacted; CP = compacted; Soil texture: S = sand, L = loam, C = clay.

The sensitivity of the water balance calculations also varies when different calculation approaches are used. The following summaries are given here only to indicate some of the general trends of the sensitivity analysis (Perrier, et al., 1980):

- Leachate generation and evapotranspiration are significantly affected by changes in the soil-water storage and the available water capacity;

- The winter cover factor is seasonally dependent and directly affects sensitivity of the evapotranspiration;

- The SCS curve number primarily affects the surface runoff and secondarily affects both the evapotranspiration and the leachate generation;

- The impermeable liner only affects water that has percolated past where there is control by evapotranspiration and surface runoff;

- The surface runoff was the most sensitive parameter when varying the barrier soil depth;

- The effects of the leaf area index are seasonally dependent and the parameters most sensitive to changes in leaf area index were evapotranspiration and leachate generation;

- The primary parameters affected by the barrier soil compaction were leachate generation, and surface runoff; and

- Changes in soil texture are highly time dependent and produce conditions where other parameters are very sensitive.

Applicability and Accuracy

The applicability and accuracy of the water balance calculations can be analyzed based on the suitability of the approaches and the closeness of the calculated results to actual measurements. Suitability of the approaches may be judged by the soundness of the theory used and the completeness, reasonableness, and sensitivity of the proposed coefficients, conditions, and calculation schemes. In general, the field measurement based techniques (e.g., on-site precipitation, surface runoff, infiltration, or evapotranspiration measurements, see Figure 25) can achieve better accuracies for the leachate volume estimation. However, these techniques are less applicable because of the difficulties involved for a large scale field measurement. Other non-field measurement based techniques are usually based on a set of assumptions or empirical data and, therefore, their use may be site or conditions specific. All non-field measurement based techniques bear the disadvantage of unknown accuracy for the landfill conditions. This is because no statistical verifications have been conducted to evaluate their use in actual landfill sites. It is very difficult to judge which non-field measurement based techniques can achieve better results for leachate estimation. Table 34 is prepared only for the qualitative comparisons among various leachate estimation methods based on the soundness and completeness of the approach, and easiness and wideness of the application.

The site verification of various leachate estimation techniques is largely lacking. There are no landfills designed specificially for the purpose of evaluating leachate generation models. Laboratory or field test cells have been used, however, by various researchers to evaluate the accuracy of the models. Wigh (1981) found that in a large field test cell (Cell No. 1) the water balance calculations* were reasonably accurate in predicting the quantity of leachate, with only 16 percent differences after 6.5 years. But, if average evapotranspiration values had been used, rather than ones computed from the actual climatic conditions experienced, the leachate predicted would have been 43 percent different than that actually collected. In other attempts, Wigh (1979 and 1981) found that after a precipitation input of 2050 mm to all cells the leachate collected varied from 213 to 2347 mm. The low and high values that occurred were explained by leakage of the test cells. The verification of the model by these test results is therefore, impossible.

Other studies, such as Fungaroli, et al. (1979), and Walsh et al. (1981) also attempted to balance the leachate generation and input water in refuse test cells. In general, the water balance method is applicable and accurate for the test cell conditions, because, in such study designs, the difficult-to-estimate factors such as surface runoff and evapotranspiration, are usually not involved.

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* Following assumptions were used for calculation:
- Runoff coefficients--0.17 and 0.13 for wet and dry seasons, respectively;
- Soil moisture storage--125 mm for final cover (2 ft) and 90 mm required to bring moisture content of final cover to field capacity;
- Refuse field capacity--330 mm/m refuse;
- Evapotranspiration--computed from the actual climatic conditions.

TABLE 34. COMPARISONS AMONG LEACHATE ESTIMATION METHODS

Factor	Methods	Advantages	Disadvantages*
Precipitation	Existing Records	• Data are easily obtainable	• Data may not be suitable for a specific landfill site.
	Field Measurement	• Data are more accurate than that from a nearby weather station.	• It requires precipitation gauges and long-term testing to gather necessary data.
Surface Runoff	Surface Measurement	• Data are more reliable for specific landfill site conditions.	• Test plots are necessary for obtaining the data; • Field testing increases the labor and time requirements for this method.
	Rational Method	• This method is widely used by hydrologists and sanitary engineers. It is a well established reliable procedure for most engineering cases; • It is very simple to use; • Required inputs such as precipitation, soil types, surface slope, and vegetal cover, are available or can be estimated.	• A number of assumptions cannot be readily satisfied under actual circumstances; • Runoff coefficients for various landfill conditions are limited; • Certain important information such as duration and frequency of precipitation, antecedent soil moisture content, and compaction of landfill cover material are not considered in the model.

(Continued)

TABLE 34. (Continued)

Factor	Methods	Advantages	Disadvantages
Surface Runoff	Curve Number Method	• This method is widely used by agricultural engineers. It is a well established reliable procedure for predicting direct surface runoff on agricultural land; • It is easy to use but more complicated than the rational method; • Required inputs such as precipitation, soil types, land cover, and antecedent moisture condition are available or can be estimated.	• It is difficult to determine conditions of the vegetation; • Since the method is designed for agricultural purposes, the curve number obtained from the table may not be suitable for landfills; • For same types of land use and soil categories, the curve number may cover the full range of values. This means the estimated leachate volume may be significantly different depending on the subjective selection of the conditions.
	Cook's Method	• It is easy to use; • Required inputs such as slope, soil type, vegetal cover, surface depression can be estimated.	• Similar to the rational method. Used for estimating the peak flow only; • Applicability has not been proved for landfill conditions.
Infiltration	Infiltrometer Measurements	• Data are more reliable than other methods which are based on assumptions and calculations only.	• Rainfall simulators or flooding infiltrometers are required to gather the data; • Field testing increases the labor and time requirements.
	Estimation From Surface Runoff	• Same as the selected surface runoff method.	• Same as the selected surface runoff method.

(Continued)

TABLE 34. (Continued)

Factor	Methods	Advantages	Disadvantages
Infiltration	ASCE Method	• It is easy to use; • Required inputs can be estimated.	• Antecedent precipitation is not considered in the method; • Only three groups of soil type are used which increase the uncertainty of assumed infiltration capacity. Therefore, results may be very subjective.
	Snowmelt Infiltration	• Both the degree-day or U.S. Army Corps of Engineers' equations are easy to use. • In most situations, only temperature data are required.	• No field verification has been conducted for various landfill conditions. The accuracy of this method is unknown; • In most situation, the effects of wind, solar radiation, etc. (refer to Figure 1) are not considered in this method.
Evapotranspiration	Soil Moisture Measurements	• Data are more reliable than other methods for evapotranspiration estimation.	• Requires field sampling and laboratory testing or field in situ moisture measurement equipment.
	Lysimeter Measurements	• It can simulate the evapotranspiration for a wide variety of landfill conditions.	• Lysimeter may not closely simulate the field conditions, therefore, the results may not be reliable when applied to landfill areas; • Requires lysimeter, vegetation growing time, and labor to conduct the tests.

(Continued)

TABLE 34. (Continued)

Factor	Methods	Advantages	Disadvantages
Evapotranspiration	Pan Evaporation Adjusted	• It is very easy to use; • Data are readily available.	• It is not sensitive to differences between land uses, vegetation types, and soil moisture conditions; • Landfill conditions may be significantly different from free water evaporation. No field verification has been conducted to identify the accuracy of this method.
	Hedke Equation	• The equation is very simple and easy to use.	• Data of consumptive-use coefficients and minimum growing temperatures are usually unavailable; • Factors of wind, humidity, cover material, soil moisture content, etc. are not considered.
	Lowry-Johnson Equation	• The equation is very simple and easy to use.	• Data of minimum growing temperatures are usually unavailable; • Effects of types and density of vegetation, wind, humidity, cover material, soil moisture content, etc. are not considered.
	Blaney-Morin Equation	• It is easy to use; • Required inputs can be estimated.	• Effects of wind, and soil moisture content are not considered.

(Continued)

TABLE 34. (Concluded)

Factor	Methods	*Advantages	Disadvantages
Evapotranspiration	Blaney-Criddle Equation	• Similar to Blaney-Morin Equation; • Required inputs can be estimated.	• Effects of humidity, wind, and soil moisture content are not considered.
	Thornthwaite Equation	• It is a well established and widely used method; • It is easy to use; • Required inputs can be estimated; • Both potential and actual evapotranspirations can be estimated.	• The equation is based largely on experience in the central and eastern United States; • Effects of wind, and humidity are not considered; • It requires tedious calculations.
	Penman Equation	• The equation is based on the most complete theoretical approach; • It is widely used in England, Australia and eastern United States; • Required inputs can be estimated.	• It requires tedious calculations.
	Thornthwaite Tables	• Same as Thornthwaite Equation.	• Same as Thornthwaite Equation.

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* The accuracy of non-field measurement based methods for simulating field conditions has not been established. No statistical verifications have been conducted for such methods under various landfill conditions.

The difficulty involved in the verification of the applicability and accuracy of the water balance model for actual landfills is the lack of accurate leachate generation data. It is virtually impossible to collect all the leachate generated from an existing landfill if such a landfill was not designed for the purpose of obtaining all the leachate generated. SCS Engineers (1976) have conducted leachate collection studies for five existing municipal landfill sites, representing five different geographical areas and site conditions. These sites are selected for the evaluation of leachate generation models, because no other data for full scale leachate collection are available. Descriptions of the sites and actual data on leachate generation, precipitation, and evaporation are given in Appendix B. The comparison of method accuracy based on these five sites is informative and not to be used as guidance for screening or choosing any method discussed previously. In this evaluation, because necessary data were not available, only twenty-five individual methods were evaluated, as shown in Table 35. Results are shown in Table 36. Among the 125 individual cases (25 methods x 5 sites) studied, 54 cases (about 43 percent) are underestimated and 71 cases are overestimated for leachate generation. Results of 21 cases have errors less than 100 percent (see Table 36). No one method can estimate leachate generation with errors less than 100 percent for all the sites studied. In general, Cases 7, 13, 4, 10, 6, 12, 23, and 11 (in the order of overall accuracy) can achieve better leachate volume estimation for the selected sites.

It should be noted that the above described comparisons are based on limited field data. Part of the errors may in fact, result from field monitoring (SCS Engineers, 1976). It is suggested that the selection of applicable leachate volume estimation methods be based on the specific site circumstances, availability of data, scientific and engineering judgements, and the experience of the designer and operator in leachate generation estimation.

TABLE 35. METHODS USED FOR THE EVALUATION OF LEACHATE GENERATION.

Case Number	Precipitation*	Surface Runoff	Infiltration	Evapotranspiration
1	Field measurement	Rational method (ASCE Table: Table 3)	Estimation from Surface runoff	Pan evaporation adjusted
2.	Field measurement	Rational method (ASCE Table: Table 3)	Estimation from surface runoff	Blaney-Morin Equation
3.	Field measurement	Rational method (ASCE Table: Table 3)	Estimation from Surface runoff	Blaney-Criddle Equation
4.	Field measurement	Rational method (ASCE Table: Table 3)	Estimation from Surface runoff	Thornthwaite Equation
5.	Field measurement	Rational method (ASCE Table: Table 3)	Estimation from Surface runoff	Thornthwaite Table
6.	Field measurement	Rational method (Perry's Table: Table 5)	Estimation from Surface runoff	Pan evaporation adjusted.
7.	Field measurement	Rational method (Perry's Table: Table 5)	Estimation from Surface runoff	Blaney-Morin Equation
8.	Field measurement	Rational method (Perry's Table: Table 5)	Estimation from Surface runoff	Blaney-Criddle Equation
9.	Field measurement	Rational method (Perry's Table: Table 5)	Estimation from Surface runoff	Thornthwaite Equation
10.	Field measurement	Rational method (Perry's Table: Table 5)	Estimation from Surface runoff	Thornthwaite Table
11.	Field measurement	Rational method (Salvato Table: Table 6)	Estimation from Surface runoff	Pan evaporation adjusted
12.	Field measurement	Rational method (Salvato Table: Table 6)	Estimation from Surface runoff	Blaney-Morin Equation
13.	Field measurement	Rational method (Salvato Table: Table 6)	Estimation from Surface runoff	Blaney-Criddle Equation
14.	Field measurement	Rational method (Salvato Table: Table 6)	Estimation from Surface runoff	Thornthwaite Equation
15.	Field measurement	Rational method (Salvato Table: Table 6)	Estimation from Surface runoff	Thornthwaite Table
16.	Field measurement	Curve number method	Estimation from Surface runoff	Pan evaporation adjusted
17.	Field measurement	Curve number method	Estimation from Surface runoff	Blaney-Morin Equation
18.	Field measurement	Curve number method	Estimation from Surface runoff	Blaney-Criddle Equation
19.	Field measurement	Curve number method	Estimation from Surface runoff	Thornthwaite Equation
20.	Field measurement	Curve number method	Estimation from Surface runoff	Thornthwaite Table
21.	Field measurement	———	ASCE Method	Pan evaporation Adjusted
22.	Field measurement	———	ASCE Method	Blaney-Morin Equation
23.	Field measurement	———	ASCE Method	Blaney-Criddle Equation
24.	Field measurement	———	ASCE Method	Thornthwaite Equation
25.	Field measurement	———	ASCE Method	Thornthwaite Table

*Data from nearby airport precipitation gauges

TABLE 36. COMPARISONS OF LEACHATE GENERATION BETWEEN MEASURED AND CALCULATED RESULTS.

Case No.	Site A $L_{calculated}$+	Site A %difference*	Site B $L_{calculated}$	Site B %difference	Site C $L_{calculated}$	Site C %difference	Site D $L_{calculated}$	Site D %difference	Site E $L_{calculated}$	Site E %difference	Average Error %
1	4.39	362.11	22.12	1655.55	11.16	63.58	7.07	2518.52	7.88	964.86	1113
2	9.16	864.21	16.39	1200.79	6.86	444.44	8.17	2925.93	6.69	804.05	1248
3	2.86	201.06	11.14	784.13	0	-100.0	0	-100.0	2.41	225.68	202
4	0	-100.0	7.32	480.95	0	-100.0	.15	-44.44	0	-100.0	165
5	10.07	960.0	18.6	1376.19	7.78	14.24	4.98	1744.44	7.13	863.51	992
6	0.63	-33.68	7.81	519.84	4.02	219.05	1.21	348.15	1.8	143.24	253
7	1.56	64.21	2.92	131.75	0	-100.0	0	-100.0	0.6	-18.92	83
8	0	-100.0	0	-100.0	0	-100.0	0	-100.0	0	-100.0	100
9	0	-100.0	0	-100.0	0	-100.0	0	-100.0	0	-100.0	100
10	2.43	155.78	6.46	412.70	0.88	-87.08	0	-100.0	1.74	135.14	178
11	2.2	131.58	12.29	875.39	4.56	-33.04	2.94	988.89	1.13	52.70	416
12	3.77	296.84	7.4	487.30	1.88	-72.39	1.64	507.41	0	-100.0	293
13	0.64	-32.63	3.15	150.0	0	-100.0	0	-100.0	0	-100.0	97
14	0	-100.0	0	-100.0	0	-100.0	0	-100.0	0	-100.0	100
15	4.68	392.63	10.43	727.78	1.42	-79.16	0.77	185.19	1.08	45.95	286
16	4.8	405.26	24.82	1869.84	9.2	35.10	8.47	3037.04	4.08	451.35	1150
17	9.2	868.42	18.67	1381.75	4.61	-32.31	9.46	3403.70	2.89	290.54	1195
18	3.47	265.26	13.43	965.87	0	-100.0	0	-100.0	0	-100.0	306
19	0	-100.0	10.76	753.97	0	-100.0	0.6	122.22	0	-100.0	247
20	10.11	964.21	21.61	1615.08	6.17	-9.40	5.29	1859.26	4.07	450.0	980
21	13.11	1280.0	13.2	947.62	6.9	1.32	14.82	5388.89	0	-100.0	1543
22	15.91	1574.74	7.66	507.94	1.42	-79.15	14.21	5162.96	0	-100.0	1485
23	1.82	91.58	1.6	26.98	0	-100.0	4.89	1711.11	0	-100.0	406
24	0	-100.0	0	-100.0	0	-100.0	3.46	1181.48	0	-100.0	316
25	9.8	931.58	9.92	687.30	3.76	-44.93	13.44	4877.78	0	-100.0	1328

*% difference = $\frac{\text{calculated - measured}}{\text{measured}}$ x 100%; Measured Leachate: Site A = 0.95, Site B = 1.26, Site C = 6.81, Site D = 0.27, Site E = 0.74, unit in inch.

+ Unit in inch.

SECTION 4

LEACHATE COMPOSITION

INTRODUCTION

The composition of municipal landfill leachate has been the subject of considerable research during the past two decades. Landfill leachates are, in many cases, highly contaminated and can degrade surface and ground-water resources. In recent years, research efforts have shifted from merely characterizing leachate components toward identifying the solid waste and landfill factors influencing the composition of leachate, and developing models which describe the concentration histories of leachate. Though the behavior of landfills and leachate generation is generally understood, present leachate composition models are useful only in the interpretation of experimental results, rather than finding application to field-scale problems.

In this section, the chemical and microbiological character of municipal landfill leachate will be reviewed, and the influence of various factors which interact to control leachate quality will be described. In addition, the effects on leachates of adding municipal and industrial sludges to municipal landfills will be identified. Finally, leachate composition modeling efforts will be reviewed.

CHEMICAL COMPOSITION

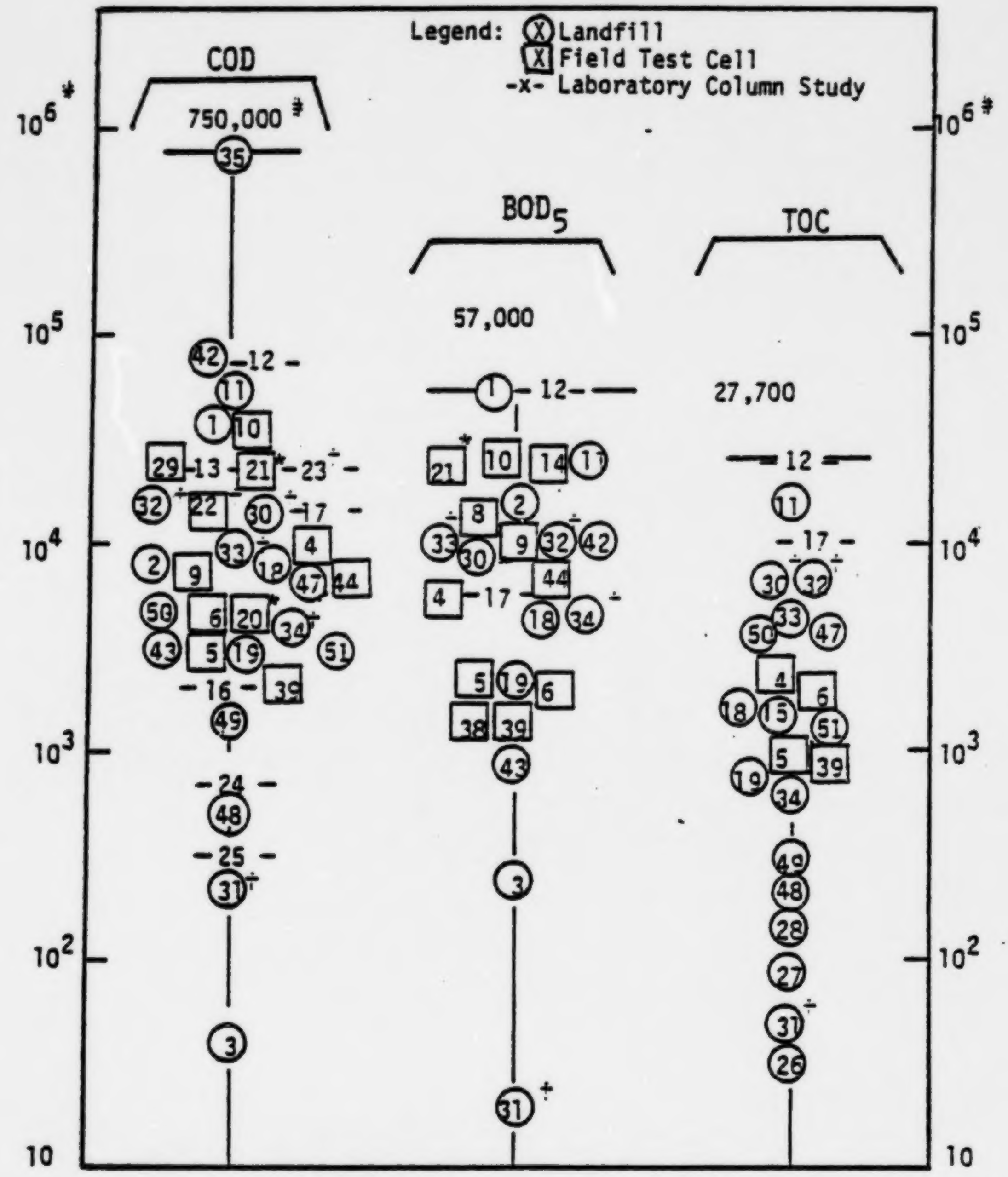
Several researchers have investigated the quality of landfill leachate under controlled laboratory conditions and in the field. Merz (1954) found that ion concentrations in leachates from percolation bins reached a maximum of 10 to 20 times the concentration found in the applied water; ammonia nitrogen and phosphorus were present in extremely high concentrations relative to the applied water. Qasim and Burchinal (1970) indicated from their lysimeter tests that leachate was high in dissolved solids and found BOD concentrations 40 to 85 times higher in leachate than in most raw domestic sewage sludge. Fungaroli (1971) reported that the mean pH of leachate was 5.5. Ranges of other constituents included: zinc, 0 to 135 mg/l; chloride, 50 to 2,400 mg/l; sodium, 100 to 4,000 mg/l; hardness, 300 to 6,000 mg/l; COD, 1,000 to over 5,000 mg/l. Hughes, et al.,

(1971) studied four landfills in northeastern Illinois and concluded that leachate quality was highly variable. Leachate contained larger pollutant loads than raw sewage or industrial waste; concentrations of heavy metals were higher than the applicable drinking water standards.

Data from these and other studies addressing the chemical composition of municipal landfill leachates were combined to provide the ranges of constituent concentrations illustrated in Figure 27. Sources of the leachate data are given by numbers indicated within each range. The location of each number in the range indicates the concentration reported by each study, or, if a range of concentrations was reported, the values indicate median values (except where noted otherwise). A legend is presented at the end of Figure 27. Appendix C summarizes the leachate composition data used to develop Figure 27.

Note in Figure 27 that data represent leachates collected from actual landfill sites, field test cells, and laboratory column tests. The sampling of leachates from actual landfill sites was in most cases achieved through monitoring wells placed in and beneath the fill. A few samples represent ground water collected downstream from the landfill sites, or leachate springs which could be sampled at the surface. Leachate samples collected above ground (data points 30,31,32,33,34) may contain eroded soil and may have reacted with the soil to significantly affect its quality. Many leachate studies have involved field test cells and laboratory columns which simulate actual landfills. A study by Wigh (1979) showed a lack of statistical similarity between leachates of small-scale cells and leachates of a field-scale cell, though the comparison was prejudiced by possible leakage caused by faulty cell construction. In studying Figure 27, no specific trends are evident which link higher or lower leachate concentrations to specific testing methods.

Figure 27 shows large variations in leachate constituent concentrations. The variations are attributable to a myriad of interacting factors, including refuse age, the rate of water application, refuse moisture, landfill design and operations, and the interaction of leachate with its environment. The variations can also be attributed to sampling procedures, sample preservation, handling and storage, and analytical methods used to characterize the leachates. For example, the effective pore diameter of the material through which leachate is collected can affect the concentration of parameters such as suspended solids, phosphates and heavy metals since precipitation or coatings may be formed on the material which may filter out these materials (Chian and DeWalle, 1975). Chian and DeWalle (1977) found rapid changes in leachate redox potential (Eh), turbidity, suspended solids and color immediately after sampling. Cook (1966) showed that leachate stored in a quart jar capped



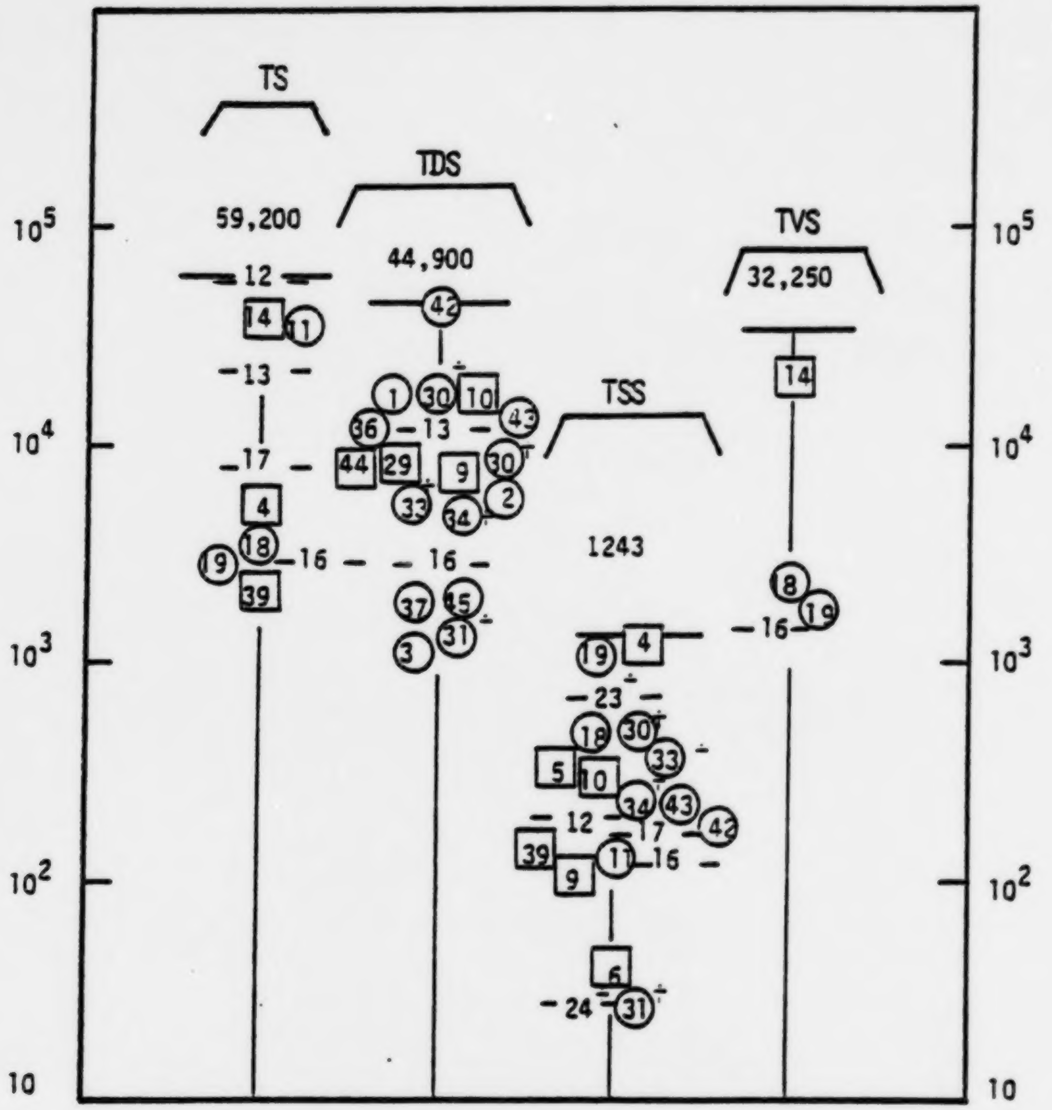
* denotes peak values

÷ denotes arithmetic mean concentration

# units in mg/l unless indicated otherwise

** denotes maximum reported value (mg/l)

Figure 27. Leachate constituent concentration ranges.



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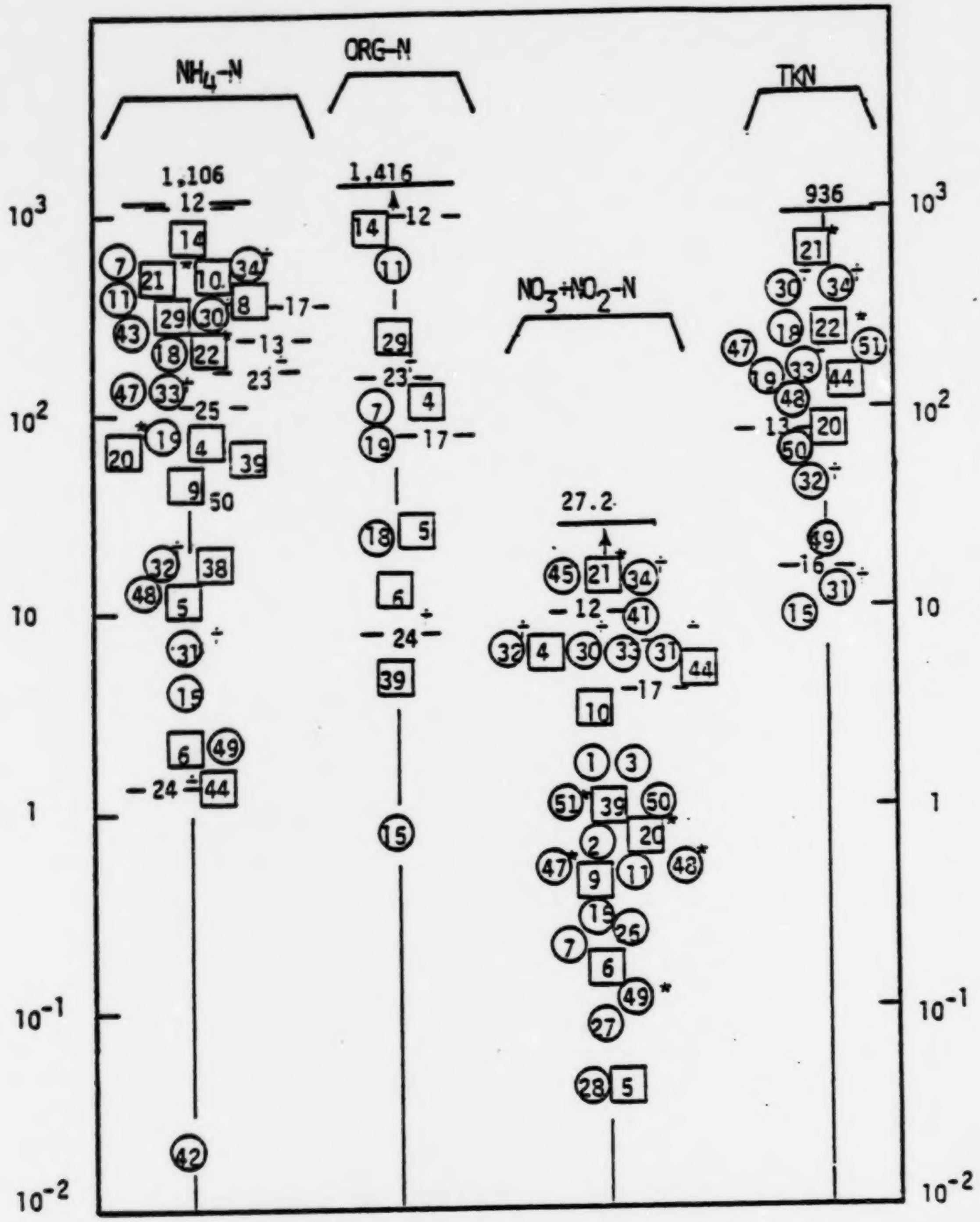

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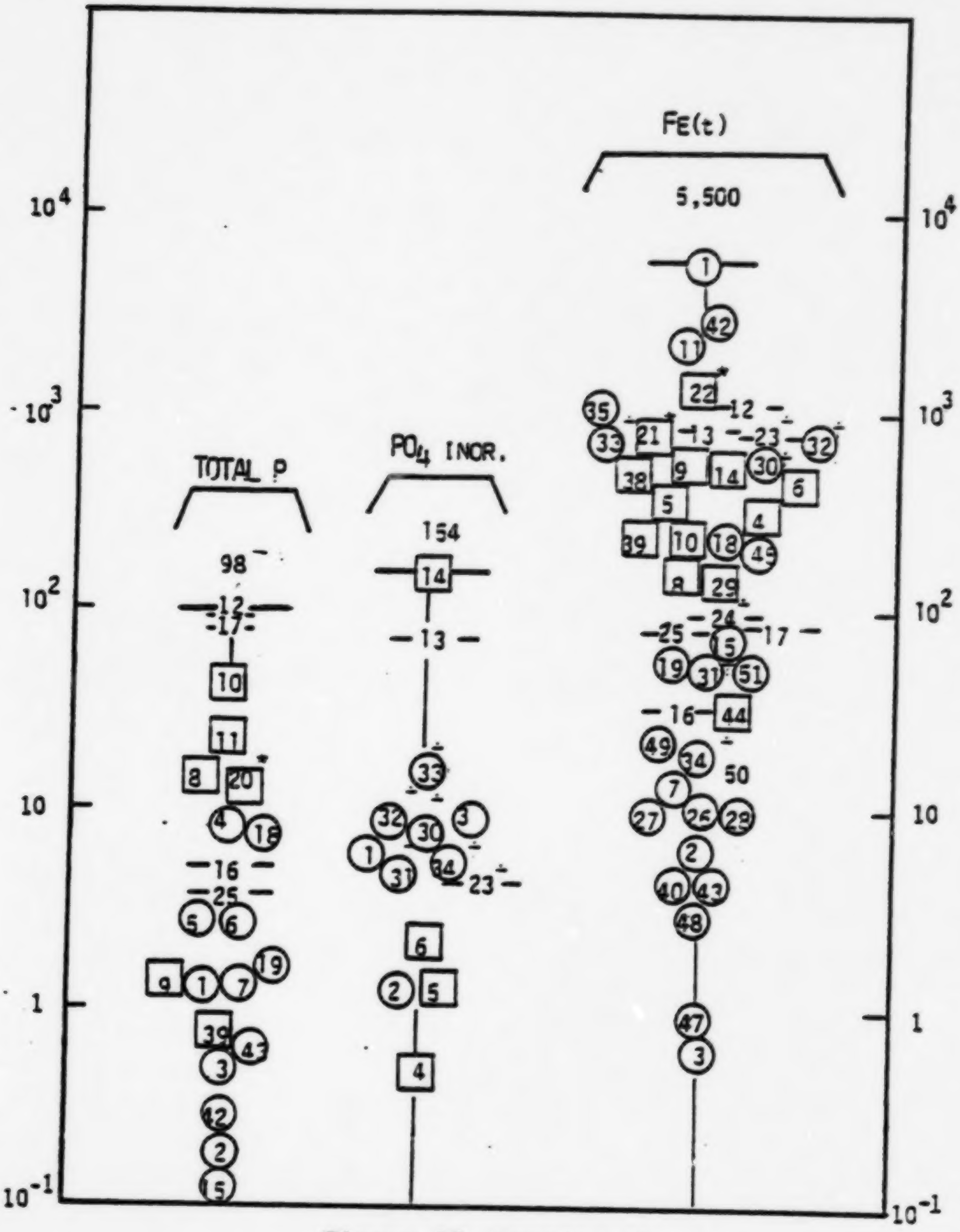

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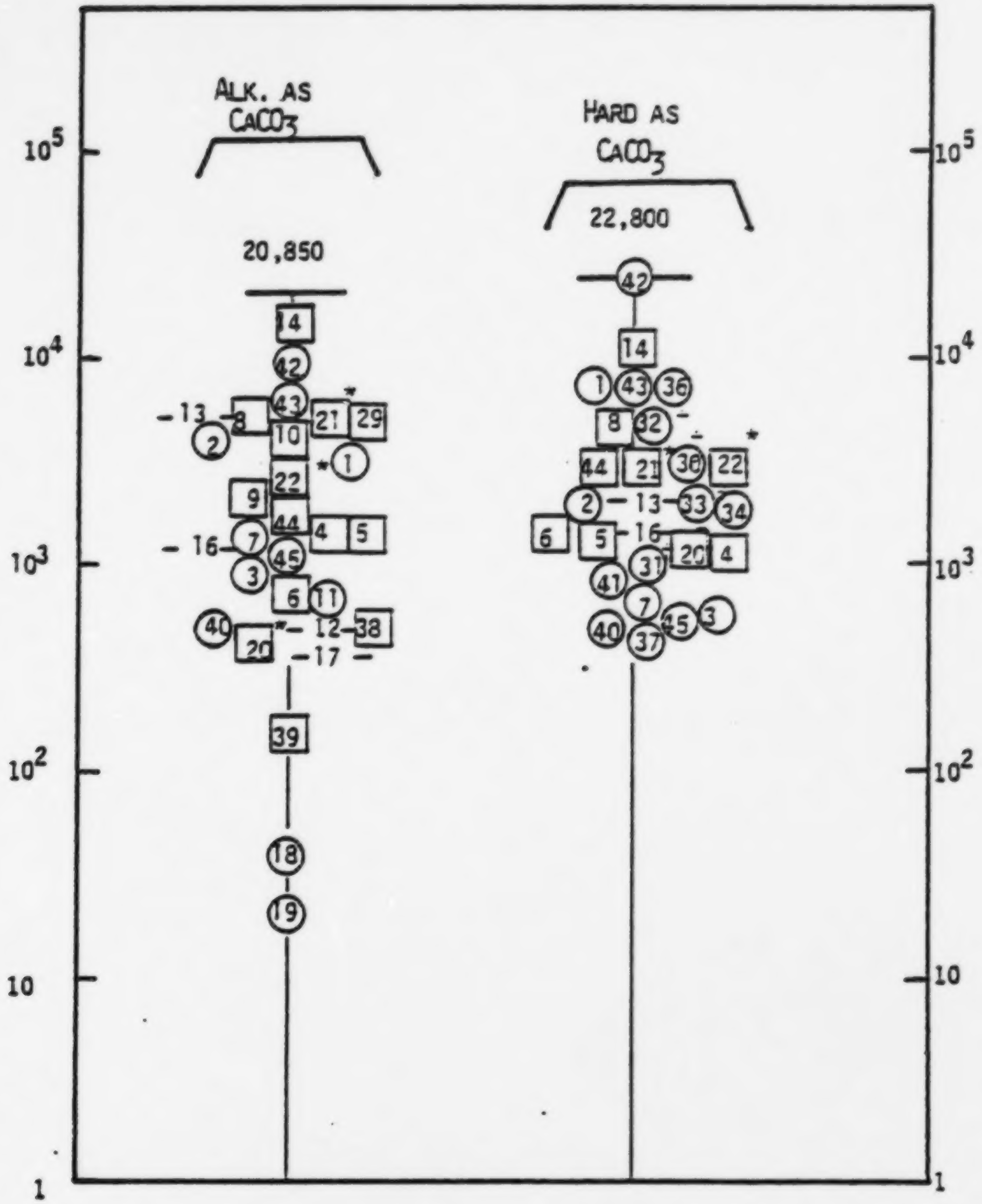

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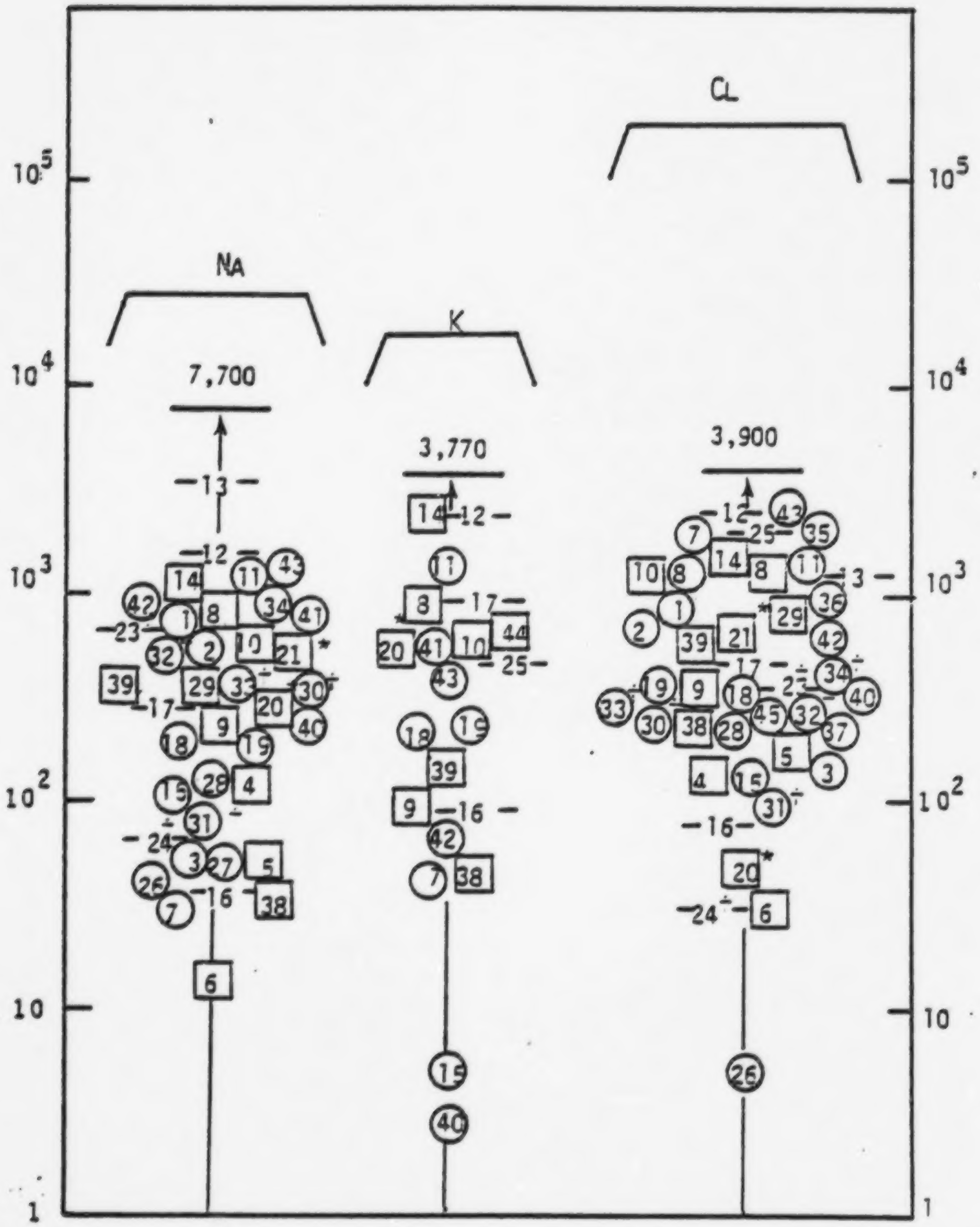

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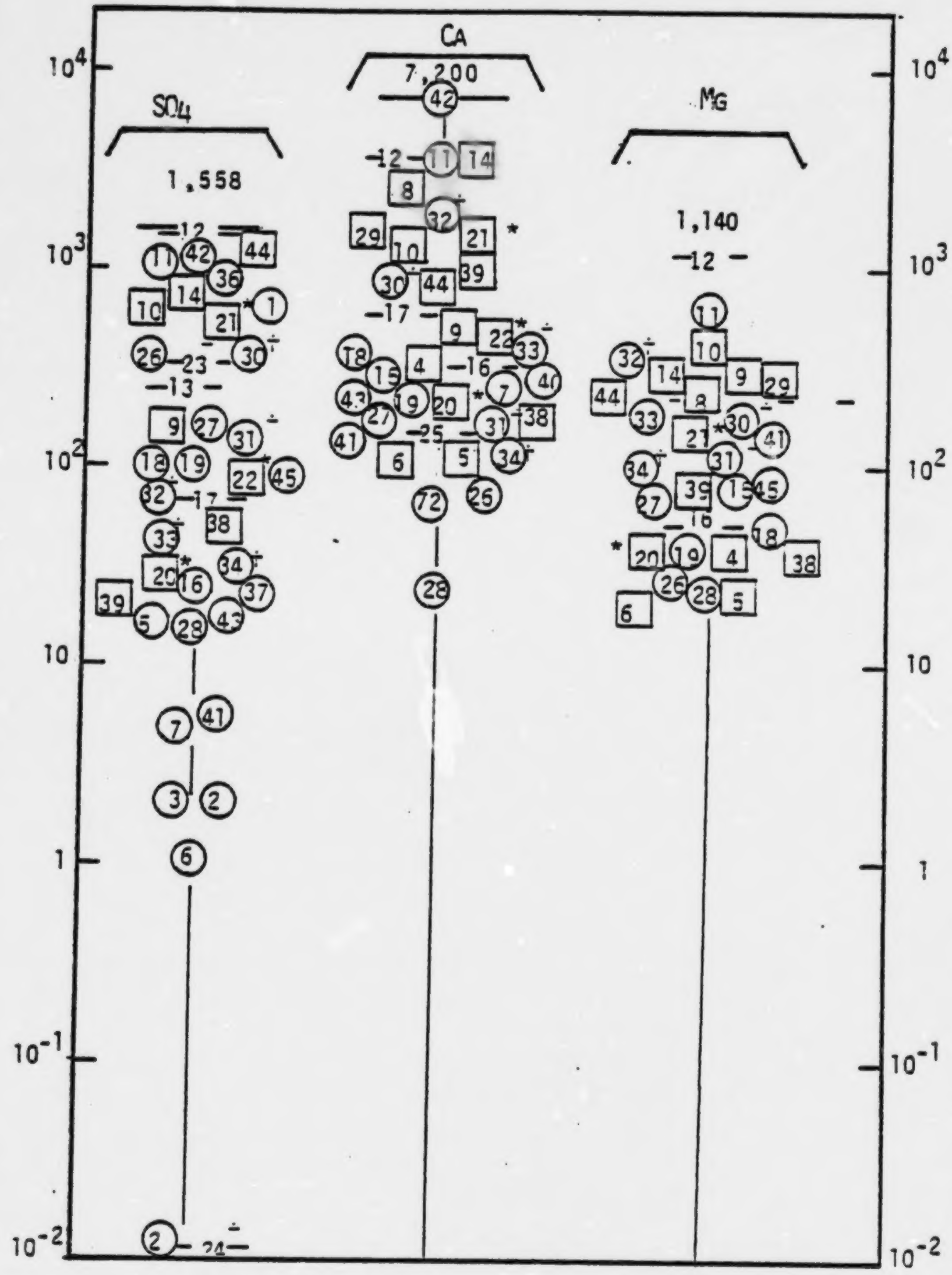

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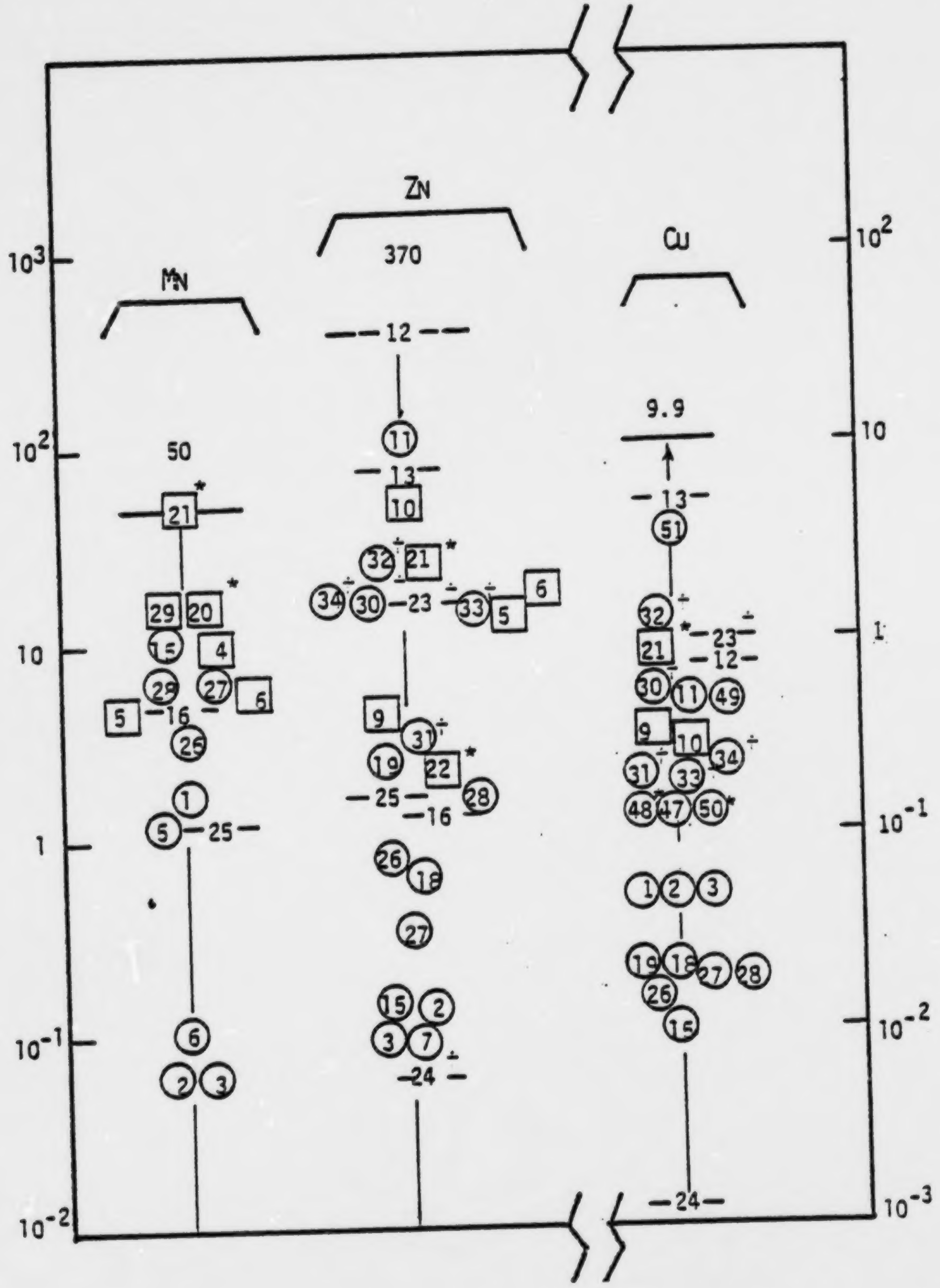

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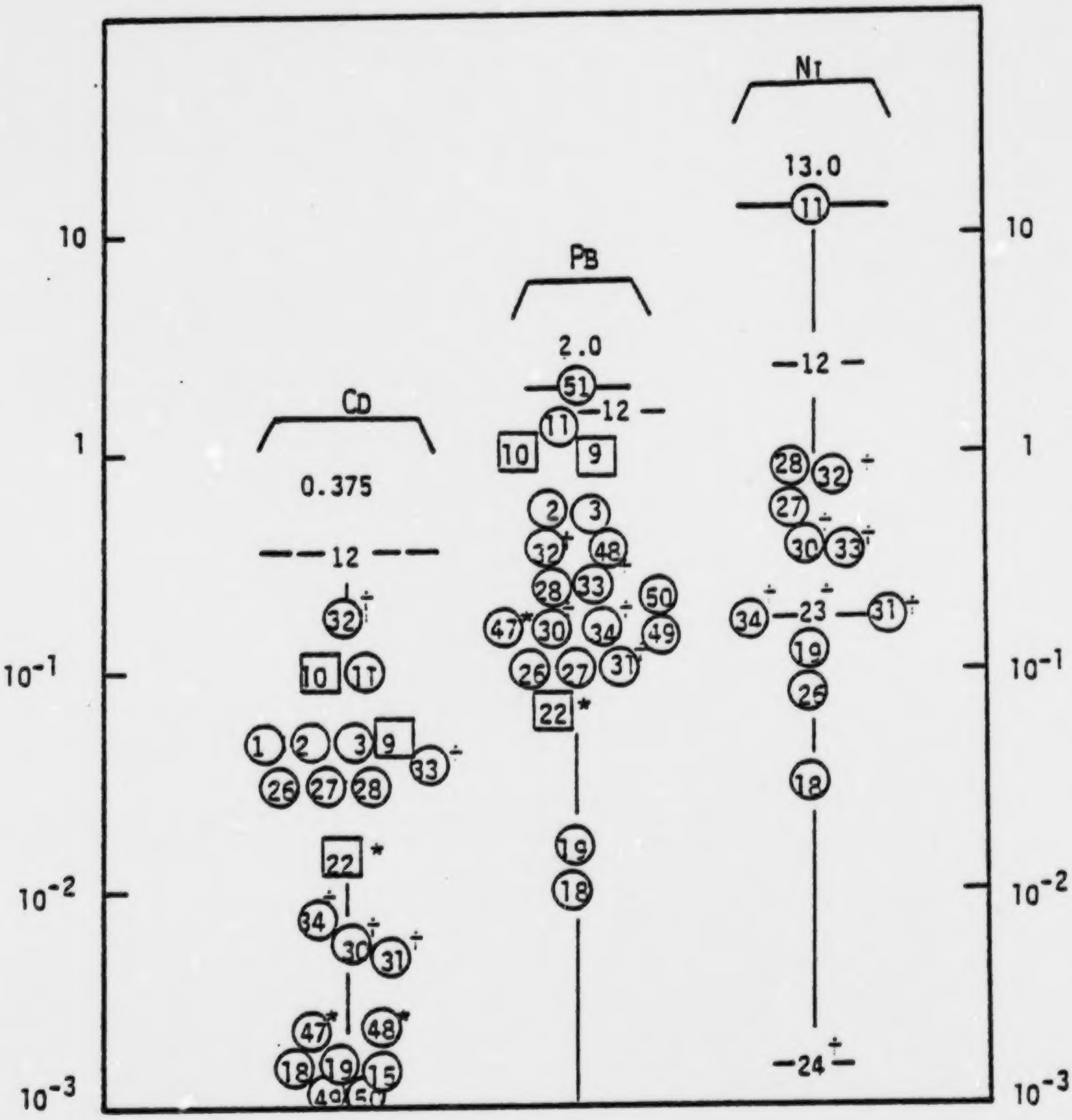

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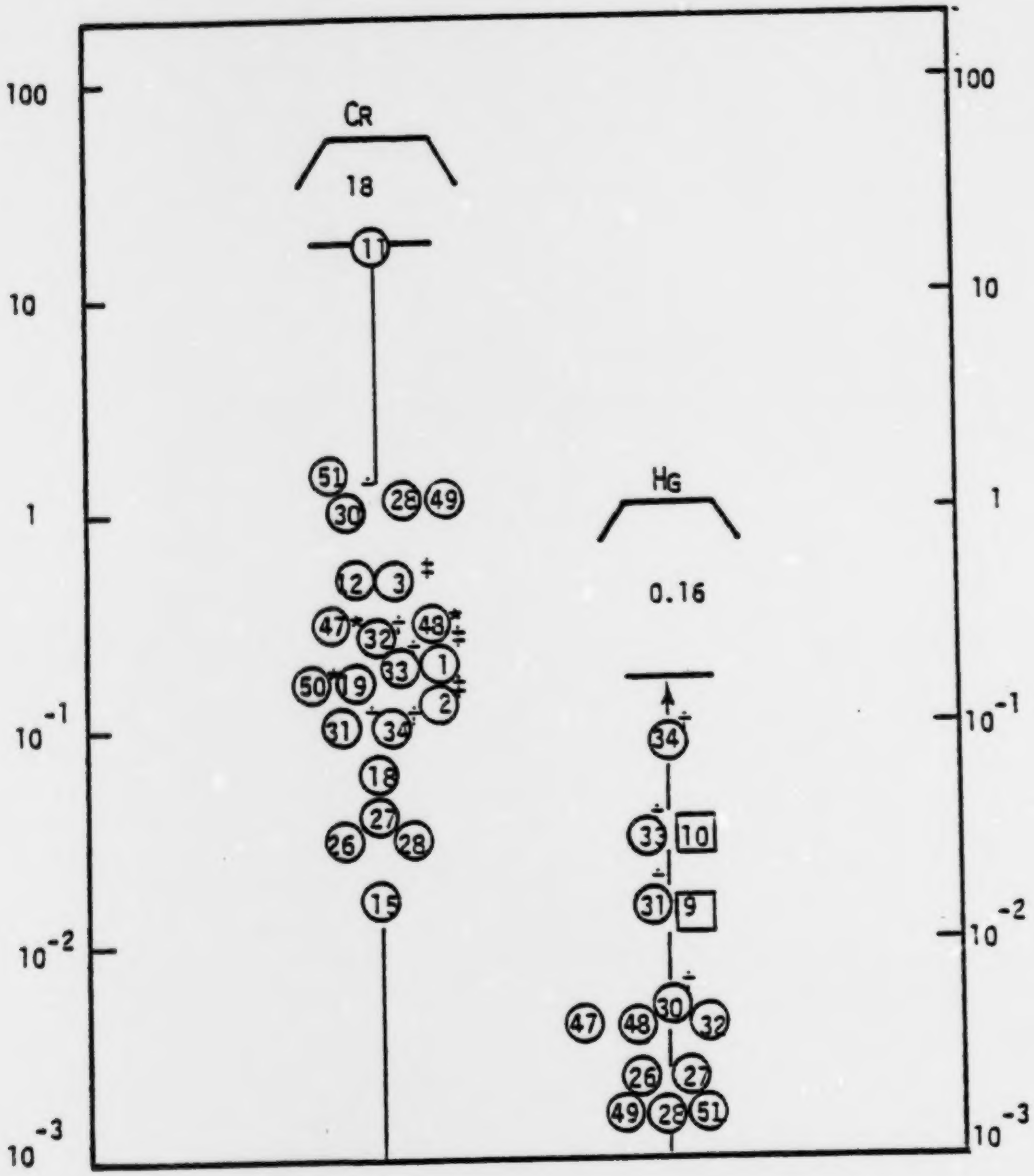

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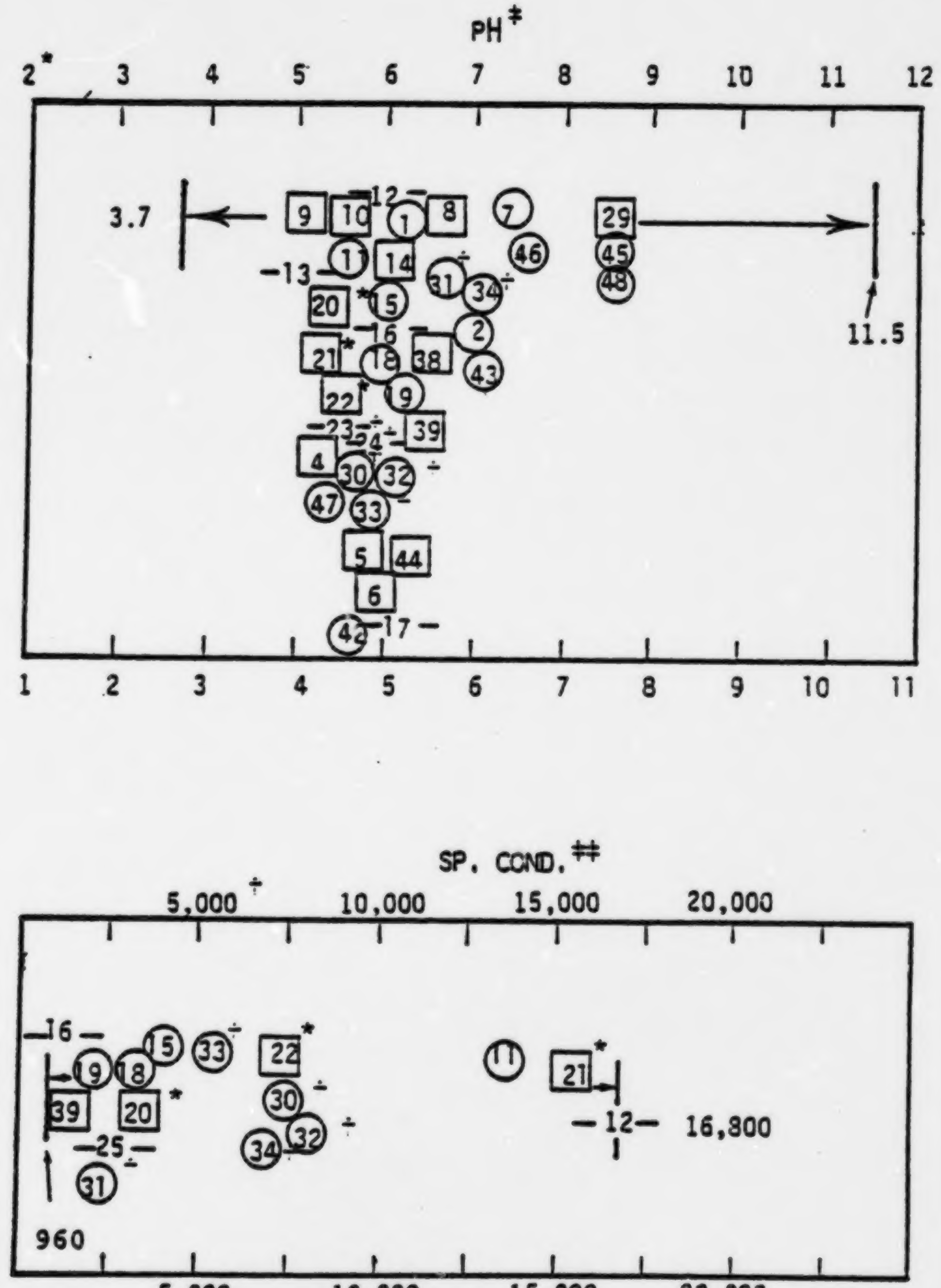

‡ in pH units

in µmhos/ml

Figure 27 (Continued)

Where:

Number(s)		Reference
1-3	indicates	Hughes, et al. (1971)
4-6	"	Pohland (1975)
7	"	Ministry of Local Housing and Government (1961)
8	"	Merz (1954)
9,10	"	EMCON Associates (1974)
11,12	"	Chian and DeWalle (1977)
13	"	Fungaroli (1971)
14	"	Qasim and Burchinal (1970)
15	"	Zenone (1975)
16	"	Chun (1975)
17	"	Reinhart and Ham (1973)
18,19	"	Johansen and Carlson (1976)
20-22	"	Wigh (1979)
23,24	"	Fungaroli and Steiner (1979)
25	"	Fuller (1978)
26-28	"	U.S. Army Engineer W.E.S. (1978)
29	"	Rovers and Farquhar (1973)
30-34	"	SCS Engineers (1976)
35-37	"	Emrich and Landon (1969)
38-39	"	Apgar and Langmuir (1971)
40	"	Anderson and Dornbush (1967)
41	"	Riccio and Hyde (1971)
42,43	"	Meichtry (1971)
44,45	"	County of Los Angeles (1969)
46	"	Zenone (1974)
47-51	"	SCS Engineers (1978)

Figure 27 (Concluded)

with aluminum foil and maintained at room temperature caused a 55 percent COD reduction after three weeks with most of the decrease occurring after one week. Most studies, however, fail to mention leachate sampling, preservation, handling, storage, and analytical procedures, making interpretation and comparison with other studies difficult.

Aside from the leachate constituents presented in Figure 27, several classes of organic compounds have been identified in municipal landfill leachates. Those compounds identified by Burrows, et al. (1975), Khare and Dondero (1977), Robertson, et al., (1974), and Engers (1977) are summarized in Table 37. Short chain fatty acids, indicative of the primary stages of anaerobic degradation, are common to all investigations. Research has revealed that organic constituents in landfill leachate are largely a factor of landfill age (Chian and DeWalle, 1977). Further discussion of organic constituents is presented in a later subsection entitled "Factors Affecting Leachate Composition," (Landfill Age).

TABLE 37. ORGANIC COMPOUNDS OR CLASSES IDENTIFIED IN LANDFILL LEACHATES

AUTHOR	COMPOUND
Burrows & Rowe (1975)	Acetone Short chain alcohols Short chain acids
Khare & Dondero (1977)	Alkanes Ketones (Acetone, 2-butanone) $CHCl_3$, CCl_4 Aromatic solvents (benzene, toluene, xylene) Short chain alcohols Short chain amines Short chain acids
Robertson, Toussaint, & Jorqne (1974)	Phthalate esters Aromtaic solvents (cresol, xylene) Toluo ethyl-toluene sulfonamide Alcohols Methylpyridine Ethers Short chain acids
Engers (1977)	Short chain acids (C_1 - C_7)

MICROBIOLOGICAL COMPOSITION

Municipal solid waste contains a large microbial population, and may be heavily contaminated with pathogenic microorganisms (Gaby, 1975). According to Gaby, refuse is an excellent medium for supporting the survival of pathogens. Municipal solid waste landfills often contain pet feces, animal remains, disposable baby diapers, hospital wastes, and sometimes sewage sludges, all of which pose a potentially significant health hazard. Although the chemical properties of municipal landfill leachates have been fairly well-characterized, the microbiological composition and viability of contained microbes has been less studied. Microbiological investigations have centered on the detection of fecal-indicator bacteria which, if found, would suggest the presence of microbial pathogens in municipal leachates. Various bacteria, and fecal-indicator groups, viruses, fungi, and parasites identified in municipal landfill leachates are described below.

Bacteria

Several studies have shown that there can be a significant bacterial population associated with municipal landfill leachates. These studies have also indicated that this population varies with landfill or refuse age: population densities are high in leachates of fresh refuse but decrease with age or time of leaching (Qasim and Burchinal, 1970; Blannon and Peterson, 1974; Engelbrecht and Amirhor, 1975; Glotzbecker and Novello, 1975). Specific bacteria and fecal-indicator bacteria identified in municipal landfill leachates are listed in Table 38, along with reported numbers. Note the high upper range for the three fecal-indicator bacterial groups. The presence of the fecal streptococcal group indicates fecal contamination from humans or chickens (Skinner and Squesnel,1978). Specific fecal streptococcal species identified in municipal landfill leachate are _Streptococcus_ _faecalis_ biotype III and _S_. _equinus_ (Table 38). The presence of the fecal coliform group indicates fecal contamination by warm blooded animals. This group is readily identified by its ability to ferment lactose, producing gas at 44.5°C. The third group, total coliform, is the traditional indicator of possible pathogenic constituents in water supplies. A water or leachate analysis resulting in a positive indication of coliform bacteria implies a potential danger, though the sample can contain coliform bacteria and still not contain any pathogenic organisms.

Table 38 also shows three bacterial species marked by double asterisk. These groups are judged by the U.S. Public Health Service (1976) as Class 2, moderately infectious agents posing a potential health hazard. According to the Public Health Service, those working with Class 2 agents need to take

TABLE 38. BACTERIA AND BACTERIOLOGICAL INDICATORS IDENTIFIED IN MUNICIPAL LEACHATES

Organism or fecal-indicator group	Range of reported values	Type of Study[*]	References[+]
Indicator groups			
Fecal streptococci	0-48 million organism/100 ml. <0.1-3 million CFU/ml.[‡] <2-490,000 MPN/100 ml.[#]	lab, FTC, landfill	1,2,4,5,6, 7,8
Fecal Coliform	0-1.5 million organism/100 ml. <0.1-100,000 CFU/ml. <2-94,000 MPN/100 ml.	lab, FTC, landfill	1,2,3,4,5,6, 8,9
Total Coliform	<0.1-100,000 CFU/ml. <2-140,000 MPN/100 ml.	lab, FTC, landfill	4,5,6,7,9
Organism			
Streptococcus sp.		landfill, FTC	8,10
S. Faecalis var. *liquefaciens*		FTC	1
atypical *S. Faecalis* var.		FTC	1
S. faecalis biotype II		FTC	1
S. faecalis biotype III		FTC	1
S. durens		FTC	1
S. equinus		FTC	1
E. coli		landfill, FTC	2,10
Salmonella typhimurian	0-450,000 CFU/ml	lab, landfills	3,7
Achromobacter sp.		landfill	8,10
Bacillus sp.		landfill, FTC	8,10
Clostridium fallax		landfill	8
Clostridium sp.		landfill, FTC	8,10
Corynebacterium sp.		landfill, FTC	8
Aeromonas sp.		FTC	8
Listeria monocytogenes[**]		FTC	8
Micrococcus sp.		FTC	8
Moraxella sp.		FTC	8
Neisseria sp.		FTC	8
Serratia marcescens		landfill, FTC	8,10
Acinetobacter sp.[**]		FTC	8
Enterobacter cloacal		FTC.	8
Stpahylococcus albus		FTC	8
Pseudomonas sp.		landfill	10
Flavobacterium sp.		landfill	10
Lactobaccilus sp.		landfill	10
Sarcina sp.		landfill	10
Propionibacterium sp.		landfill	10

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[*] lab = laboratory column or lysimeter test; FTC = field test cell simulating an actual landfill; landfill = actual landfill

[+] 1. Blennon and Peterson (1974)
2. Glotzbecker and Novello (1975)
3. Engelbrecht (1973)
4. Engelbrecht (1974)
5. Cooper (1974a)
6. Qasim and Burchinal (1970)
7. Engelbrecht and Amirhor (1979)
8. Scarpino and Donnelly (1979)
9. Sobsey (1978)
10. Cook (1967)

[‡] CFU- colony forming units

[#] MPN = most probable number

[**] Agents of ordinary potential hazard such as staphlococci, that can cause disease when the agent penetrates the skin (Public Health Service, 1976)

precautions to prevent from being innoculated by contaminated material containing these pathogens.

A comprehensive review of studies on the survival of bacteria in leachates was conducted by Ware (1980). In reviewing her work, a number of trends are evident. First, increases bacterial mortality with time of leaching or refuse age result from the bacteriacidal effects of the leachate and landfill. Bacterial survival is inversely proportional to landfill temperature (Glotzbecker and Novello, 1975; Engelbrecht, 1973; Engelbrecht and Amirhor, 1975). Relatively high temperatures (60°C) achieved in the aerobic stage of refuse biodegradation (Farquhar and Rovers, 1972) can thus inhibit bacterial growth and survival. Bacterial inactivation is more rapid at lower pH (Engelbrecht, 1973; Engelbrecht and Amirhor, 1975). Together, temperature and pH can act synergistically to accelerate bacterial inactivation (Engelbrecht and Amirhor, 1975). Attempts to correlate leachate chemical and biological composition implicated iron and zinc cations and short chained fatty acids in increasing bacterial inactivation, though not all bacterial species were similarly affected (Engelbrecht and Amirhor, 1975). Riley, *et al.*(1977) found that aliphatic acids present in raw leachates exerted a bacterio-static action on coliforms.

Recent research on the health hazard associated with municipal landfill leachates by Donnelly and Scarpino (1981) has demonstrated that leachates can contain high concentrations of bacteria which are present in human feces, several strains being pathogenic for humans. Though fecal-indicator bacterial concentrations were usually low (below detection level) for landfills inactive for 2 years or greater, fecal-indicator bacteria were isolated in solid waste from a 9 year old landfill. Pathogens found in these old landfills were *Listeria monocytogenes*, *Acinetobacter* sp., *Moraxella* sp., and *Allescheria boydii*,[*] all of which are moderately infectious (U.S. Public Health Service, 1976). Donnelly and Scarpino also studied bacterial population changes in leachates from small-scale experimental lysimeters. Low coliform levels were detected in leachates after 4 months. Streptococcal levels remained higher, but eventually dropped below detection level after 1¼ years. Laboratory lysimeters containing sewage sludges generated leachates which contained fecal-indicator bacteria after 2 years.

Viruses

Because municipal solid waste may contain fecal material from a number of different sources, it is possible that enteric viruses are among the pathogens entering leachate. As opposed to bacteria, viruses are obligate parasites and cannot multiply outside a host organism (Mitchell, 1974). They may be present in the feces of humans and animals which are subsequently disposed in municipal landfills. Enteric viruses have been detected in fecally contaminated articles in municipal solid waste (Sobsey, 1978).

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[*] a pathogenic fungus

In general, enteric viruses are rarely found in municipal landfill leachates. Engelbrecht, *et al*.(1974) detected no viruses in leachates produced by a large, field scale municipal solid waste lysimeter that had been experimentally contaminated with poliovirus type 1 during the filling operation. Peterson (1971) detected poliovirus type 3 (150 PFU[*]/100 ml) in one of 13 samples of leachate collected from a lysimeter containing raw municipal solid waste. The experimental lysimeter was rapidly brought to field capacity, thus enhancing the chances of viral detection in the leachate. Likewise, Cooper, *et al*. (1974a, 1974b) only occasionally detected enteric viruses in leachates from a series of pilot scale municipal solid waste lysimeters seeded with poliovirus. The occurrence of virus in leachate was influenced by the rate at which the test units were brought to field capacity: viruses were detected in leachate when they were brought to field capacity rapidly. The investigators speculated that the sporadic appearance of viruses in leachate may have been the result of their irregular distribution in the fill or because of non-uniform flow of the water applied to the fill.

Using a method which detects enteric viruses with about 40 percent efficiency in experimentally contaminated leachates, Sobsey (1978) examined 22 leachate samples from 21 different landfills in the United States and Canada for enteric viruses. The sites represented a broad range of conditions for solid waste landfills and the leachate samples ranged from 10 to 18 liters in volume. Enteric viruses were detected in only one of the 22 leachate samples examined. Two viruses, indentified as types 1 and 3, were found in an 11.8 liter sample obtained from newly placed refuse at an improperly managed sanitary landfill site. From his studies, Sobsey (1978) concluded that leachate samples from properly operated sanitary landfills do not constitute a public health hazard due to enteric viruses. This finding was consistent with the results of previous laboratory and pilot-scale studies on the occurrence of enteric viruses in landfill leachate (Sobsey, 1975).

Municipal leachate and landfills apparently pose a harsh environment for the survival of viruses, though the mechanisms of viral destruction is presently unknown. The rate of viral inactivation in leachates is temperature-dependent and proceeds much faster at higher temperature (e.g., 20-22°C) then at lower temperatures (e.g., 4°C) (Engelbrecht and Amirhor, 1975;

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[*] Plaque forming units

Sobsey, *et al*., 1975; Novello, 1974). Elevated landfill temperatures can thus enhance the inactivation of viruses. It has been speculated that adsorption of viruses onto solid waste components and particulate matter in leachates has in part accounted for the non-detection of viruses in viral detection studies. Ware (1980) references various studies (i.e., Carlson, *et al*., 1968; Moore, *et al*., 1975; Schaub and Sagik, 1975) which have demonstrated that viruses retain their infectious nature upon adsorption.

Fungi

Very little information is available in the literature on the presence of fungi in landfill leachates. Cook, *et al*. (1967) identified many saprophytic (or nonpathogenic) molds such as *Aspergillus*, *Penicillium*, and *Fusarium* in older landfill leachates. Donnelly and Scarpino (1981) examined leachates from commercial and experimental field landfills and frequently isolated yeast cells of basically 3 or 4 genera. Also identified were the genera of *Fusarium*, *Penicillium*, *Sepedonium*, *Cephalosporium*, and *Allescheria boydii*. Except for *Allescheria boydii*, which can cause madura foot abscesses, all fungi are saprophytic. In the same study, Donnelly and Scarpino determined the numbers of fungal cells, namely yeasts and molds, in leachate obtained from six different experimental laboratory lysimeters using Sabouraud dextrose agar plates. The plates were incubated at room temperature for one week and counted for molds and yeasts. Although each lysimeter was prepared with different types of municipal waste, hospital waste, sewage sludge and combinations of any two of them, fungi cell numbers were mostly in the range of <10 to 10^4 CFU[*]/ml even 23 weeks after lysimeter construction.

Parasites

A review of literature failed to show any reference to parasites in municipal landfill leachates. Parasites, including protozoa, helminths, and nematodes are a potential constituent of municipal leachates because of the presence of human and animal feces in landfills, and particularly when unstabilized sewage sludges are received by a landfill. Parasitic cysts and ova are among the hardiest of fecal microorganisms; many landfill conditions which inactivate bacteria and viruses are ineffective for parasites, particularly nemetodes and helminths (Hays, 1977). Many parasites found in sewage sludge (and thus human feces) are capable of causing disease (Lu, *et al*., 1982). Research to determine the potential for parasites to survive in municipal landfill leachates and potential health hazards associated with parasites in leachate is needed.

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[*] Colony forming units

Summary

Research clearly indicates the presence of pathogenic bacteria in leachates of fresh refuse in a sanitary landfill. Viruses are only occasionally detected in leachates, though their potential presence in leachates of fresh refuse cannot be overlooked. There does appear to be a significant decrease in bacterial and viral populations with refuse age or time of leaching. Leachate and landfill conditions appear to present major obstacles to microbial survival in leachates. Elevated landfill temperatures resulting from biodegradation can help to inactivate bacteria and destroy virus. The chemical characteristics of leachate have also been shown to contribute toward bacterial inactivation. Little work has been done regarding the presence and persistence of fungi and parasites in leachate.

PRINCIPAL MECHANISMS LEADING TO TRANSFER OF REFUSE MASS TO THE PERCOLATING WATER

Principal mechanisms by which the refuse mass is transferred to percolating water can be divided into three major categories. One major mechanism is the entrainment of refuse particulate and soluble material by the percolating water. The other two major mechanisms are the dissolution of soluble salts in refuse and stabilization of the refuse, that is, the conversion of biodegradable organic material to gaseous and soluble forms. The latter two mechanisms may be referred to as refuse solubilization processes, as solid refuse matter is converted to soluble form. Since refuse solubilization processes can play a major role in influencing the quality of landfill leachate, it is useful to summarize the general sequence of refuse stabilization and to identify those mechanisms leading to the transfer of refuse mass to the percolating water at each stage of stabilization.

Municipal solid waste stabilization in a landfill can be separated into two major biological stages: an aerobic decomposition phase, and an anaerobic decomposition phase which develops once the oxygen originally present in the landfill is consumed. The transformation from the aerobic phase to the anaerobic phase can occur rapidly after refuse placement, i.e., in a matter of hours (Myers _et al_., 1979), or may be postponed where oxygen persists (e.g., near the landfill surface or in air pockets*). The aerobic phase of decomposition is generally short because of the high biochemical oxygen demand (BOD) of the refuse and limited amount of oxygen present in a landfill. Leachate produced during this phase is characterized by the entrainment of particulate matter, the dissolution of highly soluble salts initially present in the landfill, and the presence of relatively small amounts of organic species from aerobic degradation. The leachate formed during this initial phase is likely a result of moisture squeezed out of the refuse during compaction and cell construction.

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* Fungaroli and Stiener (1979) showed that pockets of aerobic and anaerobic activity can exist concurrently within the refuse. Evidence also suggests that the attainment of anaerobic conditions is more rapid for refuse of high moisture (Chian and DeWalle, 1979), shredded refuse (Hentrich _et al_., 1979) and at greater landfill depth (Fungaroli and Steiner, 1979).

As the initial anaerobic biodegradation processes occur, acid fermentation will prevail, yielding a low pH leachate, high volatile acids concentrations, and considerable concentrations of inorganic ions (e.g., Cl^-, $SO_4^=$, Ca^{+2}, Mg^{+2}, Na^+). In this state the pH generally decreases due to volatile fatty acid production and the high partial pressure of CO_2. The increase in cation and anion concentrations probably result from the leaching of readily solubilized materials including those originally available in the refuse mass and those made available by biodegradation of organic matter.

Initial anaerobic biodegradation processes are carried out by a mixed anaerobic microbial population, composed of strict and facultative anaerobes. The facultative anaerobes aid in the breakdown of materials and reduce the redox potential so that methanogenic bacteria can grow.

The second stage of anaerobic biodegradation is characterized by methane fermentation via methanogenic bacteria. Methane fermentation generally begins within one year following refuse placement (Merz and Stone, 1968; Rovers and Farquhar, 1973; Chian and DeWalle, 1979; Walsh and Kinman, 1979). Methanogenic bacteria require the absence of oxygen, and tolerate only a narrow pH range (6.6 to 7.4). Nutrients must be present in a readily assimilable form. The refuse and leachate pH regulates the extent of retention of numerous soluble nutrients while the redox potential determines whether or not several inorganic nutrient elements are in an appropriate oxidation state for assimilation.

Leachate and gas compositions reflecting near neutral pH, low volatile acids concentrations, low total dissolved solids concentrations, and high CH_4 contents (>50%) in the gas are typical of older landfill sites which have already undergone substantial stabilization of the readily available organics in the refuse. Although the rate of solubilization of most constituents is significantly diminished at this point, the stabilization processes continue to work for many years. A small portion of the original refuse organic content, the lignin-type aromatic compounds, are not degraded to any extent anaerobically and remain in the fill material. These compounds are important factors in adsorption and complexation mechanisms as described below.

As mentioned above, the quality of leachate resulting from the solubilization of the refuse mass can be influenced by a variety of physical, chemical, and biological activities occurring within the landfill. These activities not only control the presence or absence of a constituent in leachate, but can alter chemical species and influence the physiology of degrading microbial populations. Each of the major activities and their importance in influencing leachate quality is briefly described below:

pH

The pH of a refuse and leachate system will influence chemical processes (precipitation, dissolution, redox reactions, and sorption) and will affect the speciation of most of the constituents in the system. In general, the acid pH conditions, characteristic of the initial anaerobic biodegradation state of refuse stabilization will (1) increase the solubilization of chemical constituents (oxide, hydroxide, and carbonate species) (2) decrease the sorptive capacity of the refuse, and (3) increase ion exchange between leachate and organic matter.

Redox Potential

The redox potential of a refuse and leachate system will affect the oxidation state and chemical form of many constituents in the system. Redox reactions are related to biological activity in the refuse, to diffusion of oxygen from the atmosphere, and to addition of oxygen from rainfall. Because of these factors and the variations in refuse local environments (or microenvironments), redox equilibrium probably cannot be reached. Reducing conditions corresponding to the second stage of anaerobic biodegradation will influence the solubility of nutrients and metallic solids, resulting in precipitation or dissolution of these constituents. Changes in redox potential can also alter sorption reactions because of solid species transformations.

Adsorption and Complexation

Adsorption and complexation are probably the most important processes influencing the attenuation or mobility of trace metal constituents by the refuse mass. Under oxidizing conditions adsorption can regulate the concentration of a constituent well below the level controlled by precipitation effects. Lignin-type aromatic compounds can adsorb trace metal constituents from the leachate, as can iron and manganese hydrous oxide solids, hydrated aluminum oxides, and clay minerals from daily or interim soil cover (Jenne, 1968). In complexation, metal ions combine with non-metallic compounds called ligands by means of coordinate-covalent bonds. Leachates are abundantly provided with such ligands as chloride, ammonia, phosphate and sulfate as well as an array of organic compounds which provide conditions under which complexation must be considered in evaluating the transport and fate of toxic metal ions (Knox and Jones, 1979; Pohland et al., 1981). In general, complexation acts to increase the concentration of metals to levels far in excess of their normal solubilities (Stumm and Morgan, 1970; Snoeyink and Jenkins, 1980). Of extreme importance, however, is the impact of sulfide solubility equilibria or the levels of trace metal complexes which can exist in the presence of sulfide (Pohland, et al., 1981). In essence, sulfide effectively competes with most complexing agents, and consequently many heavy metals will precipitate as sulfides rather than remain in solution as complexes. A comprehensive discussion of the impacts of complexation of metal solubility and the influence of sulfide is presented by Pohland et al. (1981). The net effect of adsorption and complexation mechanisms upon leachate composition is difficult to quantify because of the complex and heterogeneous nature of refuse and landfill systems.

Temperature

Temperature affects not only the solubility of many chemical constituents and the reaction rates, but it establishes limits of biological activity. Thus, temperature flux within a landfill system may affect the reaction mechanisms and biodegradation rates.

The solubilities of many inorganic salts increase with temperature (e.g., $CaPO_4$, NaCl), while a number of compounds of interest in leachate (e.g., $CaCO_3$, $CaSO_4$) decrease in solubility with increased temperature. Temperature represents a major variable on biological activity. For biological systems, each microorganism will possess an optimum growth temperature, ranging from 0°C to as high as 80°C. An increase or decrease from the optimum temperature will result in decreased growth due to such things as enzyme destruction.

In terms of reaction kinetics, an increase in temperature will increase the reaction rate. A 10°C increase in temperature will roughly double reaction rates, though exceptions include dissolution and biological reactions.

Biological Mechanisms

Throughout the life of a landfill, important biological processes regulate abiotic reactions, which in turn initiate further biological response. The effects of refuse bacteria, fungi, actinomycetes, and protozoa on the chemical states of various constituents in the refuse and leachate system are numerous. Biological activity affects oxidation and reduction, mineralization, immobilization, precipitation, and complexation. The most important factor is the effect of biological activity on the oxidation-reduction reactions and organic compound transformation.

Oxidation-reduction reactions are greatly affected by the degradation of organic compounds. The mode of degradation not only changes with the species of organisms, but it also may affect the redox potential and thus the oxidation state and chemical forms of all constituents in both solution and solid phases of the refuse.

Through the process of mineralization, trace elements, plant nutrients, organic chemicals, microbial tissues, and organic-inorganic complexes may be converted to inorganic compounds. Through biological assimilation, the inorganic nutrients and trace metals may be transformed into microbial tissues, thus biologically immobilizing these constituents. Organic complexes accumulating in leachates from microbial synthesis and degradation have the capacity to combine strongly with trace metals and other constituents. Through these reactions the constituents in refuse and leachate can be mobilized, complexed, precipitated, or sorbed.

Summary

In summary, the rate of solubilization of a refuse mass is governed by specific microbial populations, which are in turn associated with a prescribed set of chemical and physical processes occurring in the landfill. Among these are pH effects, redox effects, precipitation, ion exchange, adsorption and complexation, biological effects, physical sorption effects, and temperature. The interaction of these processes creates a highly complex and dynamic system which, in turn, creates considerable variability in the composition of leachate. Some of the basic relationships in the refuse-leachate system are summarized below:

1. Carbon dioxide and various organic acids formed during refuse stabilization will reduce the pH of leachates from relatively fresh refuse, and consequently will influence the chemical alteration process (e.g., precipitation, dissolution, sorption, etc.) of the leachate constituents and the biological activities in the landfill.

2. Biodegradable organic compounds help to deplete free oxygen in the landfill, dropping the leachate redox potential (Eh), and will thus influence the

chemical forms and solubility of various constituents in the leachate.

3. Clay minerals and oxides or hydroxides of iron, manganese, aluminum and silicate, and complex compounds such as organic chelates or inorganic anions in refuse can adsorb, desorb, or complex the constituents in leachates and influence their concentration levels.

4. Finely divided particulates in refuse can increase leachate turbidity and suspended solids.

FACTORS AFFECTING LEACHATE COMPOSITION

Leachate composition is a function of numerous factors, including those inherent in the refuse mass and landfill location, and those created by designers and site operators. When viewed as representing a mass of potential pollutants, the chemical and microbiological character of refuse is largely uncontrollable.* Similarly, ambient air temperatures and rainfall are unalterable characteristics of a landfill site. Conversely, refuse density, refuse permeability, refuse depth, and the rate of water application to the landfill surface can be regulated. Those factors which influence leachate composition require identification for purposes of model development and so that control methods and landfill design parameters can be modified for optimum results. This section discusses the effects of various factors, both controllable and uncontrollable, on leachate composition.

Refuse Composition

The mass of refuse stored in a landfill may be viewed as representing a specified mass or quantity of pollutants. That portion available for leaching is largely a function of the physicochemical character of the solid waste, rates and extent of solid waste stabilization, and the volume of infiltration to the landfill. Research has attempted to estimate the total amount of pollutants released from a given refuse mass and the maximum concentration of each species in the leachate. To do this, experiments were developed to simulate the leaching of solid waste in the landfill environment. One approach, called extraction testing, involves mixing samples of solid waste with distilled water or other solvent (e.g., solid waste leachate), and allow contaminants to desorb from the waste for a period of 16-24 hrs. Another approach involves filling laboratory columns with solid waste, applying water to the surface, and allowing the water to percolate through the waste as in an actual landfill. Column tests are usually performed for extended periods, i.e. greater than one year. The cumulative mass of contaminants leached per kg of refuse for selected column studies is indicated in Table 39, along with extraction testing results. Note that for many of the contaminants represented in Table 39, the cumulative mass removals from tests exceed the quantity identified as available by aqueous extraction testing of the refuse. This is particularly pronounced for COD, indicating that the biodegradation of cellulose and other complex organic substates cannot be simulated by extraction tests. Note also the large masses of pollutants which can leach from the refuse mass. For example, Walsh and Kinman (1981) report that 1.4 g of iron were removed per kg dry wt solid waste over a six year period, which is equivalent to approximately 1,270 g (2.8 lbs) of iron removed per ton of refuse.

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* Reduction processes such as incineration do represent distinct changes in the chemical and microbiological character of refuse.

The total mass of refuse available for leaching may also be influenced by refuse surface area and contact time between the refuse and leaching solution. For example, solid wastes which are relatively water impervious but which contain water soluble constituents may exhibit leaching of these constituents from the waste surface during initial water contact, but not from the interior of the waste. With increasing contact time between the waste and leaching solution, additional water soluble waste constituents may be leached out of the waste. However, a point of equilibrium may eventually be reached where, although water soluble constituents are still available in the waste, the solubility limit of the leaching solution is attained and no further exchange can occur. Hence, as long as the pollutant does not reach its solubility limit in the leachate, more will be removed during a given time period. The relationship between refuse surface area and leachate composition is discussed more fully in the following section.

In some instances, research has demonstrated a pronounced influence of specific refuse components on the character of leachates emanating from the refuse. For example, Kemper and Smith (1981) found that the cumulative mass of zinc leached from shredded refuse was about one order of magnitude higher than non-shredded refuse. The investigators speculated that zinc alkaline batteries present in the refuse were split open upon shredding allowing the zinc to be released and to be made available for leaching. In another study, Cooper _et al._(1974a) studied the effects of adding disposable diapers on the composition of leachate from refuse-filled lysimeters simulating sanitary landfills. Fecal streptococci and coliform densities in the leachate were extremely high, indicating widespread fecal contamination of the refuse. These studies tend to underscore the importance of refuse composition on leachate composition.

TABLE 39. CUMULATIVE MASS REMOVAL OF REFUSE POLLUTANTS BY LABORATORY COLUMNS AND MASS REMOVAL BY EXTRACTION TESTING

	Cumulative Mass Removal		Mass Extracted		
Leachate Constituent	Walsh and Kinman (1981) (6 year period)*	Wigh and Brunner (1981) (9 year period)†	Ham et al. (1978)‡	Wigh (1975)	Fungaroli and Steiner (1979)
- all values in g/kg dry wt solid waste except where noted otherwise -					
COD	95[**]	80		4.5	44
TOC	40 (19%)				
TS	57 (8%)				
TKN	2.3 (90%)			2.1	0.38
Hardness		10		15.8	
Ca				1.70	0.95
Mg		0.55	0.56	0.56	
Fe	1.4 (17%)	3.5		0.0013	0.22
K			7.6	4.20	
Na				5.55	1.32
Cl		2.2		8.56	1.53
SO_4		1.5		3.46	
Zn			0.34		0.59
Cu			0.12		0.023
Pb			0.11		
Ni	0.0019 (18%)				0.053

* approximately 4800 mm of cumulative infiltration over a 6 year period

† approximately 4500 mm of cumulative precipitation over a 9 year period

‡ using synthetic leachate

[**] cumulative mass removed as percentage in solid waste

Refuse Processing

Refuse processing refers to shredding and baling activities which represent alternatives for refuse volume reduction for landfilling,and for material and energy recovery by many municipalities. In some cases, facilities have been constructed and full-scale operations are currently underway (Savage and Trezek, 1980). The effects of refuse shredding and baling on leachate production and composition has been included in major research efforts sponsored by USEPA (Fungaroli and Steiner, 1979; Kemper and Smith, 1981), and provide the foundation for the following review.

Shredding--

Rather than just review the empirical evidence on the effects of refuse shredding on leachate production and composition, it is useful to identify how, intuitively, shredding should influence leachates, and then compare the intuitive explanations with experimental data. In this way, a more comprehensive presentation can be made to provide better insight to shredding effects.

The physical characteristics of refuse are changed after it undergoes a communition (shredding) process. The characteristic particle size is reduced at least one order of magnitude as compared to raw refuse. The smaller particle size of in-place refuse can, intuitively, result in the following effects:

1. An increase in refuse surface area and therefore greater contact of the refuse mass with the percolating water.

2. An increase in in-place refuse density. Decreased particle size allows for greater compaction.

3. A decrease in the permeability of the refuse. Smaller particles and increased density favor a reduced flow rate through the refuse.

4. An increase in the landfill field capacity and elapsed time before the first appearance of leachate. Greater in-place density should result in greater moisture retention, and subsequently delay the first appearance of leachate.

5. A higher strength leachate and greater pollutant removal. An increased contact time produced by either reduced permeability or an increased field capacity should increase concentrations of pollutants in the leachate, and refuse communition should allow greater pollutant removal.

6. Accelerated refuse decomposition. The shredding process serves to "tear" paper and textile products, exposing fibers that otherwise would be more tightly interconnected.

Research has found that most of the above theoretical effects of shredding on landfill and leachate behavior do, in fact, occur. For example, experiments performed by Fungaroli and Steiner (1979) using experimental cylinders (76 mm diameter) demonstrated that milling (shredding) of refuse increases greatly the saturated field capacity of the refuse in comparison with unmilled refuse. The increase in field capacity, however, is nonlinear and approaches a limit as particle size decreases and density increases. The field capacities reported by Fungaroli and Steiner for compacted, unshredded refuse and compacted, shredded refuse are 1380 ml/kg and 1819 ml/kg dry refuse respectively. The study by System Technology Corporation (Kemper and Smith, 1981), using buried lysimeters loaded with municipal solid wastes and subject to controlled water additions, indicated that shredding had little effect on refuse moisture retention in comparison with unshredded refuse, but that the shredded waste in their study appeared to have not yet reached field capacity. This observation is supported by the fact that measured field capacity (1225 mg/kg) was somewhat lower than those measured by Fungaroli and Steiner (1979) and Wigh (1979). Regarding the effect of shredding on the elapsed time before the first appearance of leachate, Fungaroli and Steiner (1979) demonstrated that as the milled refuse size decreased, substantial delays in first leachate appearance occurred. The experimental design consisted of a landfill with an eight (8) foot (2.44m) refuse layer and a two (2) foot (0.61 m) soil cover. Infiltration was assumed to be eighteen (18) inches (0.46 m) per year. Conversely, Kemper and Smith (1981) observed roughly similar elapsed times before first leachate appearance for shredded and unshredded refuse. Kemper and Smith point out that dry parcels within supposedly "saturated" shredded refuse mass may have some influence on their results.

With regard to permeability, Fungaroli and Steiner (1979) found results which generally agree with what our intuition predicted. As indicated in Figure 28, shredding of refuse appears to slightly decrease its saturated permeability relative to the larger, unshredded refuse. The figure does, however, illustrate the importance of refuse density on permeability: increasing refuse density decreases its permeability. However, significant relationships between saturated permeability, density, and milled refuse size could not be established.

Regarding leachate strength and cumulative pollutant removal, results from each of the aforementioned investigations are generally consistent with intuitive predictions. Using a series of mini-lysimeters containing shredded refuse of different

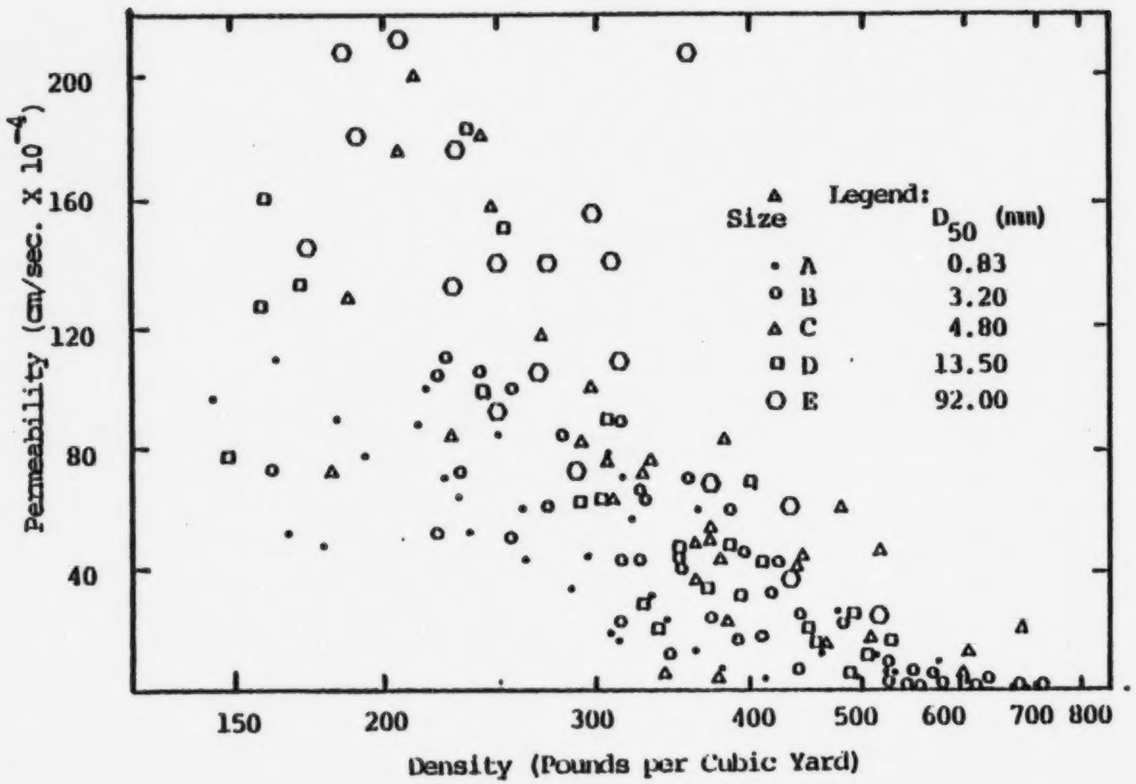


Figure 28. Relationship between refuse permeability and refuse density (Fungaroli and Steiner, 1979)

particle size, and thus refuse densities, Fungaroli and Steiner (1979) observed changes in leachate contaminant concentrations over a 76-week period. Plotting these changes with respect to refuse size and refuse density, the authors demonstrated that pollutant concentrations in leachates increased with increasing refuse density. A lysimeter with a refuse density of 737 pounds/yard^3 (437 kgs per meter^3) had the highest weekly pollutant load in 90 percent of the study period. A lysimeter with the lowest refuse density of 522 pounds/yard^3 (310 kgs per meter^3) had the lowest concentrations of pollutant in at least 75 percent of the report period. The results show an increased availability of pollutants to the leaching solution as refuse density increases. As long as the pollutant does not reach its solubility limit in the leachate, more will be removed during a given time period. Interestingly, the results also showed that except for early transient phases of leaching, particle size does not appear to have any significant influence or concentrations. Kemper and Smith (1981), however, did find that shredded refuse could form a more highly concentrated leachate in comparison with unshredded refuse. But Kemper and Smith did not attempt to differentiate between the effects of refuse density and particle size, as did Fungaroli and Steiner. Thus, refuse density may have been the ultimate cause of high constituent concentrations in shredded refuse leachates observed by Kemper and Smith. Regardless of this point, each study confirms that as a result of shredding, leachates can be of greater strength than unshredded refuse.

Kemper and Smith (1981) also showed that cumulative mass removals of leachate constituents on a basis of equal leachate volume were often greater for shredded refuse than for unshredded refuse. The authors found that the shredded refuse cell (Cell 4, Table 40) had the greatest mass removals of alkalinity, total solids, TOC, COD, iron, copper, cadmium, zinc, nickel, and lead in comparison with the remaining cells. However, the authors point out that the variability in metals results is probably caused by the large error possibilities associated with the detection of very low concentrations of copper, cadmium, nickel, chromium, and lead in the leachates. The authors attribute the higher cumulative masses removed from the shredded cell to the increased surface area of the waste and enhanced moisture infiltration, providing more sites for biological and chemical activity. With regard to the rate of pollutant removal, Fungaroli and Steiner (1979) found that milling of refuse to an effective diameter (D50) of 3.5 to 13.5 mm will significantly increase the rate of pollutant removal. Below this value, a decrease in rate of removal will occur. The maximum removal rate was reported to occur at approximately D_{50} = 10 mm.

Finally, in regard to biodegradation rates, shredding appears to promote the degradation of refuse, according to Ham and Anderson (1975). The authors noted a rapid formation of partially degraded matter in lysimeters with shredded refuse, which led to initially high pollutant loadings in leachate. Later in the study, pollutant loadings were reduced as highly degradable matter remained. The increased degradation rates were attributed to the increased homogeneity of the shredded refuse and the smaller particle sizes.

TABLE 40. CUMULATIVE MASSES OF LEACHATE PARAMETERS PER KG DRY REFUSE ON A BASIS OF EQUAL LEACHATE VOLUME (KEMPER AND SMITH, 1981)

Parameter	Cell 1*	Cell 2*	Cell 3*	Cell 4*	Cell 5*
Alkalinity, mg/kg	5,450	6,960	9,420	17,200	9,280
Total solids, mg/kg	10,500	15,200	21,200	40,200	24,200
TOC, mg/kg	6,040	9,950	13,300	21,700	16,500
TKN, mg/kg	241	411	514	712	729
COD, mg/kg	9,230	23,200	36,000	50,000	34,600
Iron, mg/kg	273	714	1,180	1,940	540
Copper, µg/kg	149	152	490	152	128
Cadmium, µg/kg	60	76	78	112	84
Zinc, mg/kg	13	20	10	200	42
Nickel, µg/kg	539	789	1,060	1,950	759
Chromium, µg/kg	246	304	305	284	167
Lead, µg/kg	598	654	564	856	577

* Cell 1. Baled, shredded refuse
Cell 2. Baled, unshredded refuse
Cell 3. Baled, unshredded refuse saturated with water
Cell 4. Shredded refuse
Cell 5. Unshredded, unbaled refuse (control)

In summary, the following trends have been confirmed by limited research with regard to the effects of shredding on leachate production and composition:

- Shredding can increase the landfill field capacity, though shredding to extremely fine sizes may increase the chances for "dry pockets" to be formed within the refuse mass, which delay attainment of field capacity;
- By increasing surface area and rates of refuse biodegradation, shredding can increase the concentration of pollutants in leachate in comparison to unshredded refuse leachates; and

- Shredding can increase both the rate of pollutant removal and the cumulative mass of pollutants released per volume of leachate in comparison to unshredded refuse.

Baling--

Municipal landfill operations, by nature, attempt to compact solid waste into the smallest volume possible to prolong landfill life. A loosely packed landfill is impractical and uneconomical. For this reason, some landfill operations have employed the preprocessing step of refuse baling. Baling has shown to have a pronounced effect upon leachate generation and quality.

Perhaps the most comprehensive study of the effects of solid waste baling on leachate generation and composition was performed by Kemper and Smith (1981). Using test lysimeters, the investigators compared the leachate produced by refuse which was processed in five different ways before placement in the cells, including a cell with baled, shredded refuse (Cell 1), baled, unshredded refuse (Cell 2), and unshredded, unbaled refuse (control, Cell 5) (Table 40). The first significant event which occurred upon water addition was that the cells with baled refuse (Cells 1 and 2) started producing leachate before the control cell (Cell 5). It was also noticed that the cells with unbaled refuse retained substantially more moisture than those containing baled refuse. The authors report that the results "indicate that baling of refuse has an inhibitory effect on its moisture retention capability. The tightly-packed bales are more difficult for water to infiltrate than loose refuse, and so absorption of water proceeds at a slower rate." The authors also note a study by Edens (1978) who also found that baled waste had absorbed little moisture one year after placement and after 8 months of leachate outflow. Analysis of the cores of the bales indicated only a 1 percent increase in moisture than what was originally present.

In terms of leachate composition, Kemper and Smith report that the leachates from baled refuse cells contained less pollutants per liter than leachates from unbaled refuse cells. Because of baling, substantially greater volumes of leachate was produced. A comparison of the mass flows of pollutants from the cells on a basis of equal leachate volume (Table 40) shows the decreasing effect of baling on refuse mass removals in comparison to non-baling. However, Cell 3 (baled, saturated refuse) has mass removals of most pollutants roughly equivalent to those of Cell 5 (unprocessed). Kemper and Smith reason that saturation of refuse in Cell 3 has obviated the inhibition of moisture infiltration normally produced by baling, thus causing greater leaching of pollutants than would be expected from baled refuse. Thus, it may be that once baled refuse has attained field capacity, the effects of baling on leachate production and composition are removed.

Returning to Table 40 , the metals results do not show the inhibition of leaching by baling of the waste since Cells 1 and 2 have mass removals approximately the same as those of Cell 5 (unprocessed). An exception is zinc which is clearly reduced in mass removals of Cells 1 and 2 in comparison with Cell 5. The variability in metals data observed, the authors note, is probably caused by the large error potential associated with the detection of low levels of metals in the leachates. Actual metal removals may actually be different than that reported.

Looking again at the cumulative masses of pollutants leached from the cells, it is interesting to compare data from Cell 1 (shredded, baled) with Cell 2 (unshredded, baled). A greater mass of every pollutant has leached from Cell 2 than from Cell 1, including over twice as much COD. The effect is clearly seen in Figure 29, showing mass removals of COD versus cumulative leachate volume. In this regard, Kemper and Smith point out that baling of shredded waste produces bales that are even more resistant to leaching than bales of unshredded waste. The shredding process evidently allows a reduction in void space within the bales, thus further retarding the movement of water in and solids out of the bales. The leaching rate from baled shredded refuse is reduced in comparison to baled, unprocessed refuse.

In review, the process of baling can enhance leachate production by decreasing the elapsed time before leaching, reducing the moisture-retention ability of the waste, and by increasing the overall volume of leachate produced. Baling of waste results in large volumes of relatively dilute leachate emanating from a landfill. Cumulative mass removals of pollutants from baled solid waste can be lower per unit volume of leachate in comparison to unbaled solid waste. However, mass removals from baled waste in a saturated environment (or saturated baled waste) were found to be similar to mass removals of saturated, unbaled waste.

Evidence has suggested that the processes of shredding and baling, taken separately, can produce opposite results on leachate generation, composition, and mass removals. Baling of waste results in large volume of dilute leachate with a longer period of stabilization than required for unbaled refuse. Conversely, shredding of waste results in the formation of smaller volumes of highly concentrated leachate, and a shorter period of waste stabilization. Large mass removal is favored by waste shredding, while smaller mass removals are favored by waste baling. The data also show that waste shredding before baling can inhibit mass removals more than baling alone. Note that all of the above-mentioned effects apply mainly to solid waste in an unsaturated state. Once field capacity has been reached for

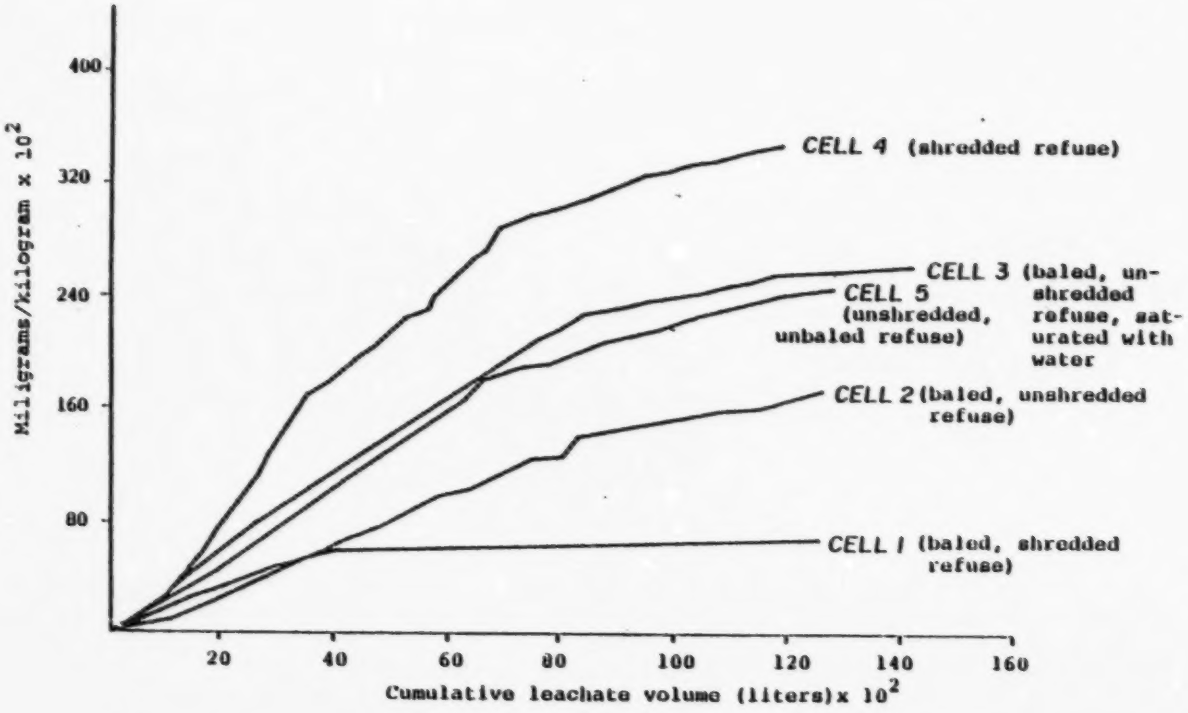


Figure 29. Mass removal of COD versus cumulative leachate volume (Kemper and Smith, 1981).

shredded, baled, or shredded/baled waste, the effects of waste processing on leachates tend to be less pronounced. It is speculated that in the long-run, cumulative mass pollutant removal per kg of solid waste will be the same regardless of waste processing, particularly for the so called non-biological parameters such as heavy metals. Solid waste and leachate constituents such as COD, TOC, volatile acids, and bacteria may be reduced in leachates through biological activity if they can be maintained in the landfill mass for prolonged periods. Further analysis of this phenomenon is covered in our discussion on leachate recycling, in Section 6, Available Control Technology.

Landfill Age

Variation in leachate composition and cumulative mass removal of pollutants in solid waste is often attributed to age factors, such as time since refuse placement or time since the first appearance of leachate. Time will obviously play an importnat role in determining leachate constituents and concentrations due to the nature of solid waste stabilization processes, described earlier. Note that landfill age, in itself, does not govern leachate compositional changes, rather it is the rates of solid waste stabilization and the rate and volume of water infiltration to the landfill. Age is merely a convenient means of measuring and monitoring changes in leachate composition and pollutant removal from waste. The literature is replete with data describing leachate quality as a function of time, some of which is summarized below.

Merz (1954) examined the quality of leachate from two percolation bins containing 3.1 m (10 ft) of compacted municipal refuse. The concentration of organic components were high in the first samples of leachate emanating from the bins and increased for five weeks. Initial BOD was 33,100 mg/l and remained high for eight months. An 80 percent decrease in BOD occurred after eight months; after 13 months the BOD had been reduced to 375 mg/l. Similarly, Fungaroli and Steiner (1979) examined leachates from an insulated lysimeter over a 6-year period. Initially high concentrations of COD (50,000 mg/l) gave way to lower concentrations (between 20,000 and 22,000 mg/l) after 2 years, and finally dropped below 5,000 mg/l after approximately 3.2 years. Most other leachate constituents exhibited concentration decreases over a 3 to 5 year period, including iron,zinc, phosphate, chloride, sodium, copper, organic nitrogen, total solids, and suspended solids. The steady decreases were attributed to the continued flushing of the refuse, thereby removing the easily decomposible and soluble materials. Operating field scale test cells with controlled moisture additions, Emcon Associates (1974) found that the refuse cell subject to continual flushing with water produced steadily declining concentrations of leachate constituents over a two-year period. These constituents included all of the general organic indicator groups, (e.g., COD, BOD) nutrients, major ions, fecal coliform, and fecal streptococcal groups. Results of leaching studies performed by Pohland (1975), using 3-foot diameter columns containing 10 feet of compacted refuse and 2.5 feet of cover soil, also revealed declining concentrations of leachate constituents, including BOD, COD, free volatile fatty acids, nutrients, and various major ions over a 3-year period. However, significant concentrations of iron (400 mg/l), and total hardness (1000 mg/l), were observed in leachate at the end of the three-year study.

Additional studies, including those conducted by Qasim and Burchinal (1970), Ham and Anderson (1975), Wigh (1979), and Walsh and Kinman (1981), have demonstrated a general trend: leachate concentration peaks for leachate constituents early in the landfill (or simulated landfill) life, i.e., within 2-3 years refuse placement, and gradual declines in ensuing years. According to literature, this trend generally applies to the organic constituents and general organic indicators (e.g., BOD, COD, TOC, etc.), nutrients, major ions, and microbiological populations of municipal solid waste leachates. In some cases, the leaching behavior of heavy metals has been characterized by this trend, but more commonly, heavy metal concentrations have fluctuated widely throughout the study periods. The inconsistency of heavy metal contaminants may be attributable to precipitation, dissolution, adsorption and complexation mechanisms which can retain or mobilize the metals within landfill microenvironments.

Figure 30 was developed in an effort to summarize the relationship between landfill age and municipal refuse leachate composition. Data used to develop the figure was derived from laboratory column studies, field test cells, and actual landfill sites; the data is compiled in Appendix C. For many of the leachate constituents, upper concentration boundaries are indicated. The boundaries were developed using a first order rate equation which, in most cases, conveniently explained the behavior of the upper limits of the experimental data. Using BOD_5 as an example, rate coefficients were calculated as follows:

$$\frac{dC_b}{dt} = -kC_b \quad \text{(first order rate equation)}$$

$$\therefore \quad C_b = C_{b_o} \, 10^{-kt} \text{ ------------------------ } (34)$$

where:

C_b = remaining (i.e., actual) concentration present at time t

C_{b_o} = maximum concentration (assume t = 0)

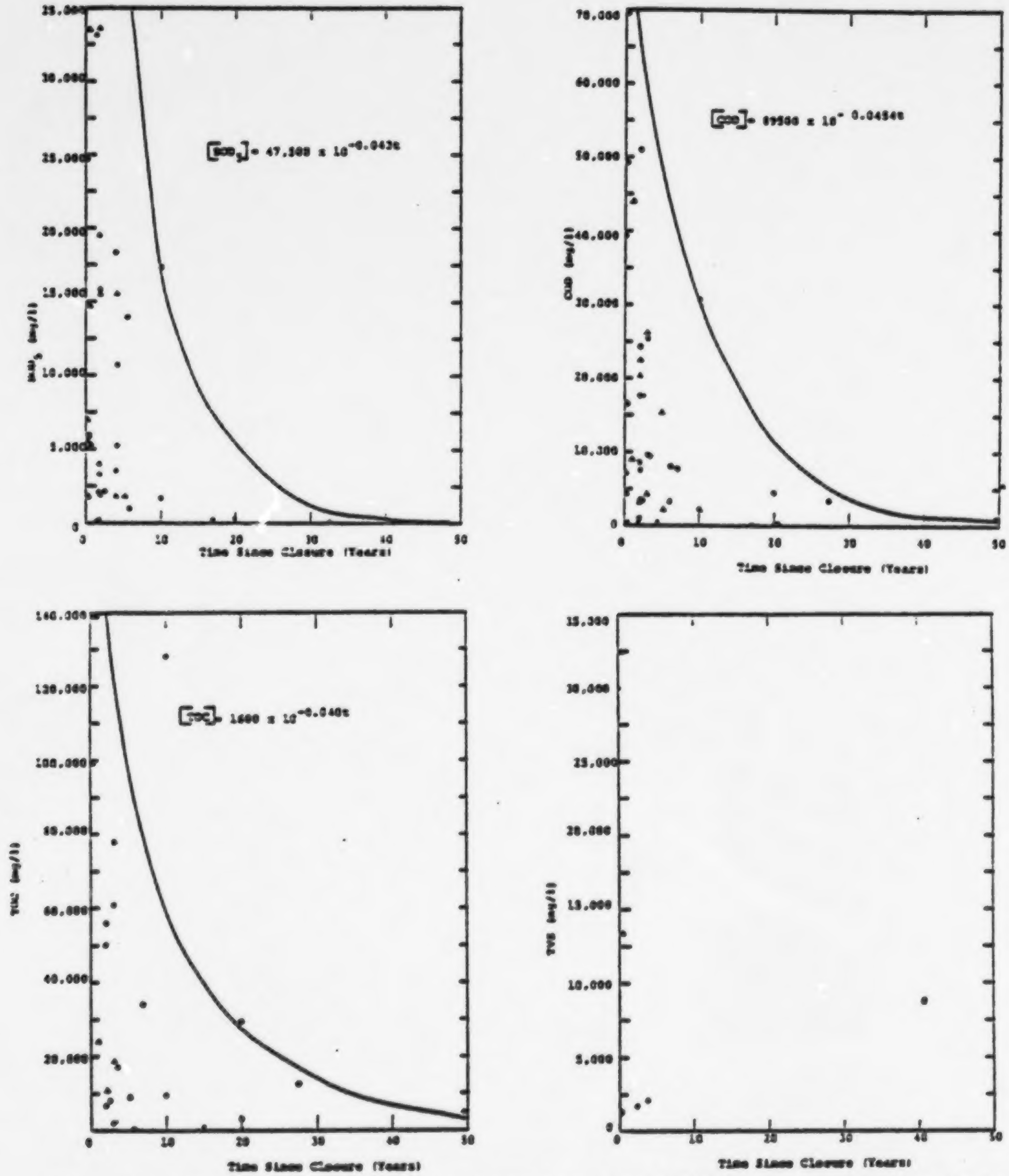


Figure 30. Relationship between landfill age and leachate composition. (Note: solid circles represent actual landfills, triangles - field test cells, and hollow circles are laboratory columns)

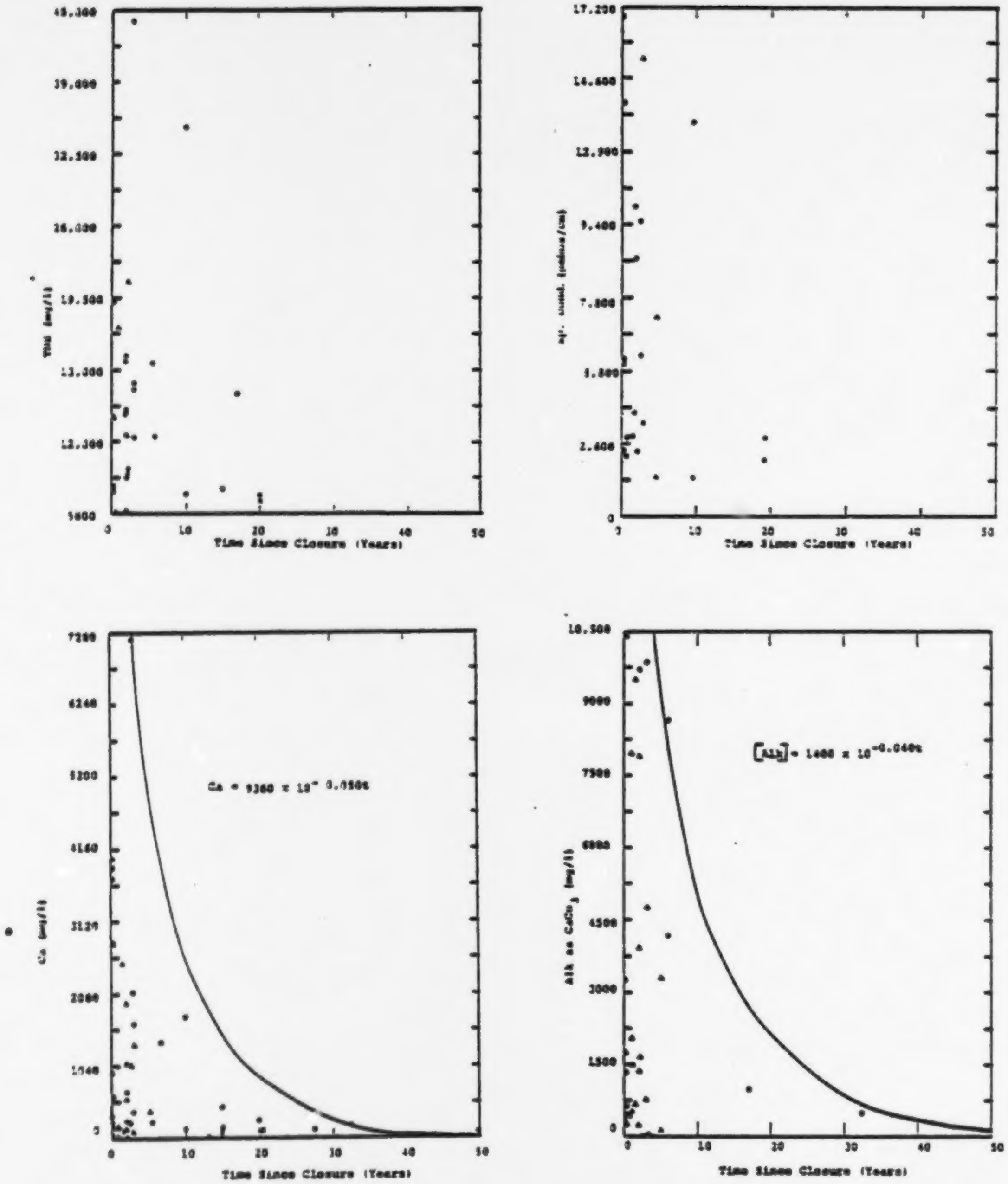
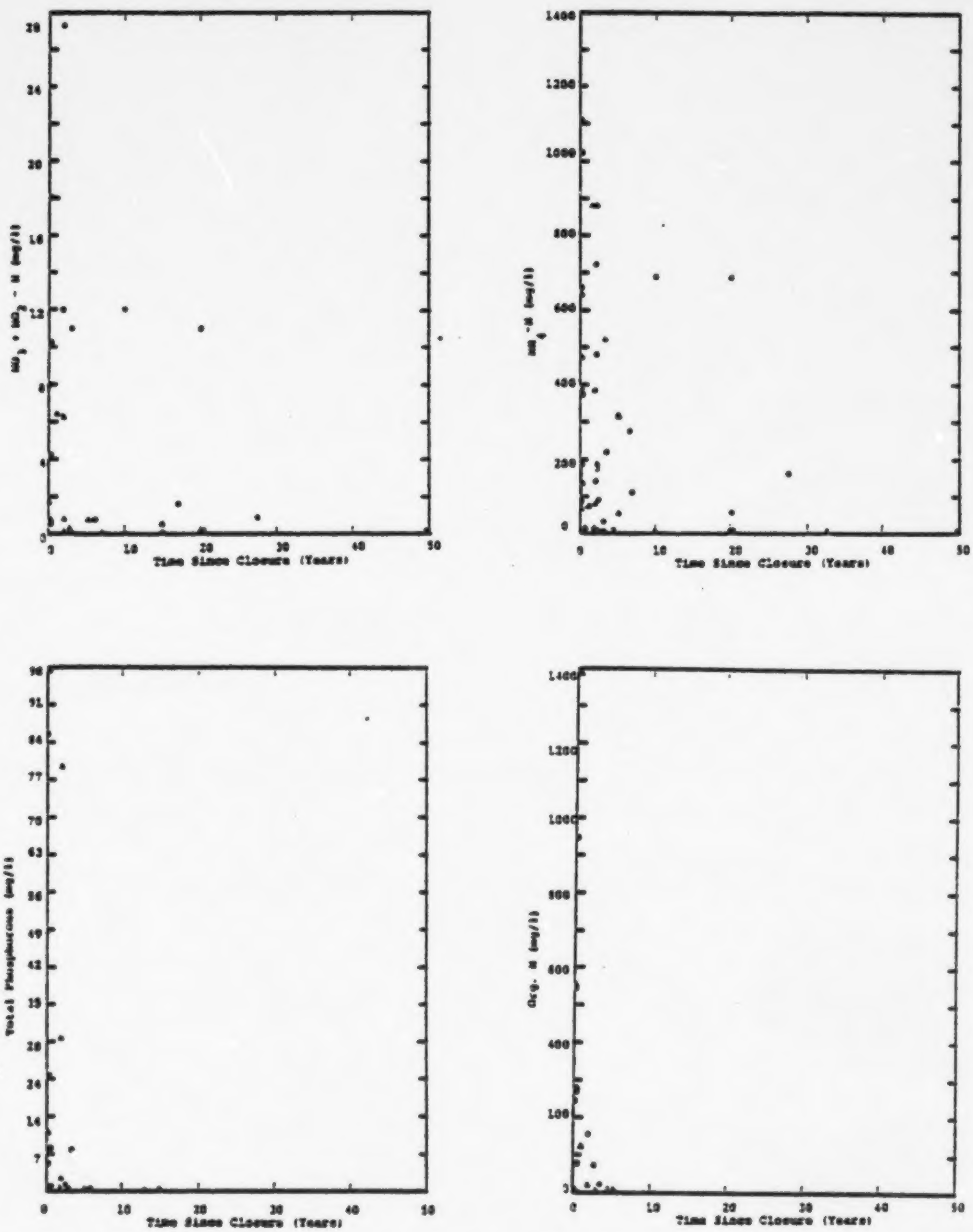


Figure 30. (Continued)



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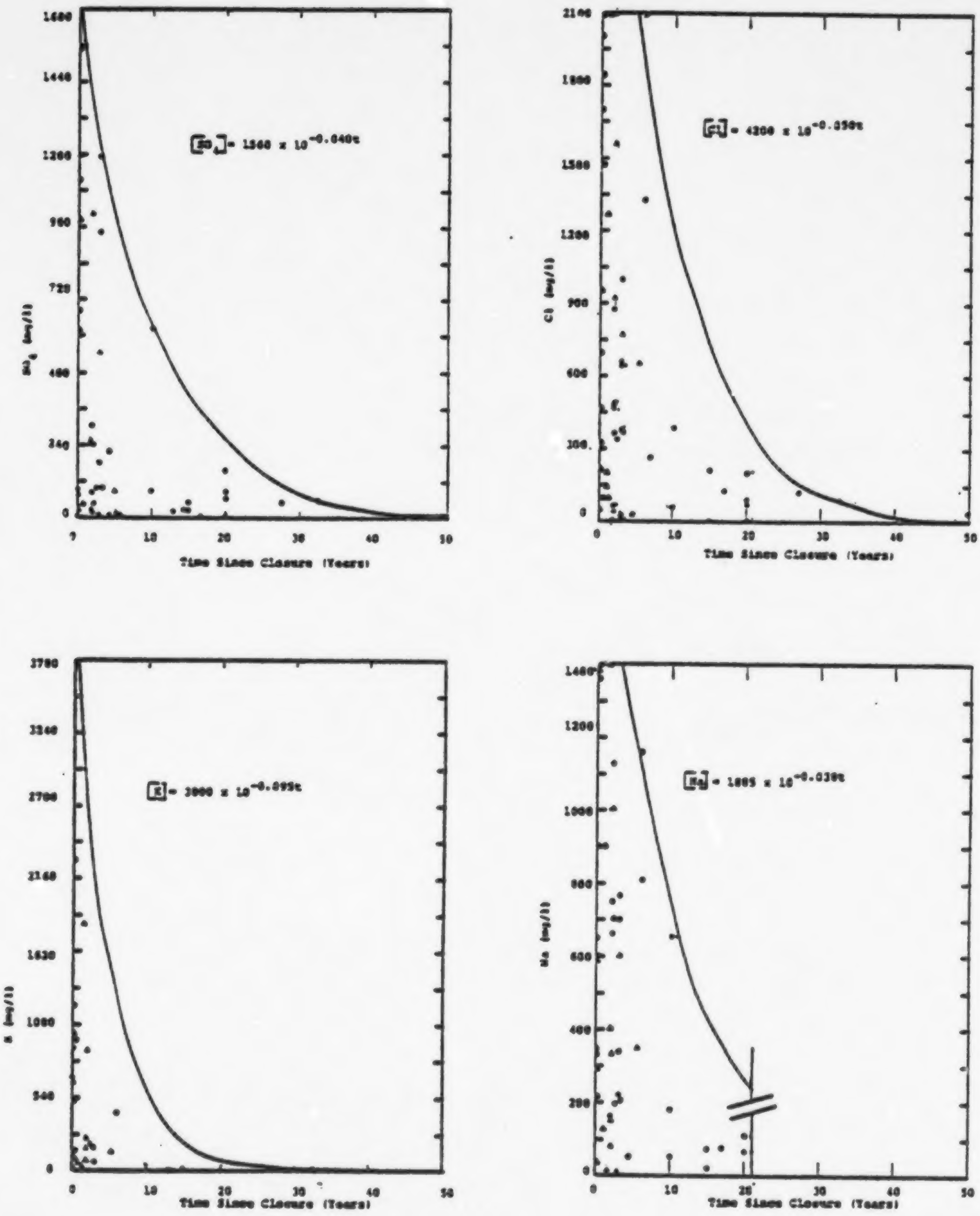

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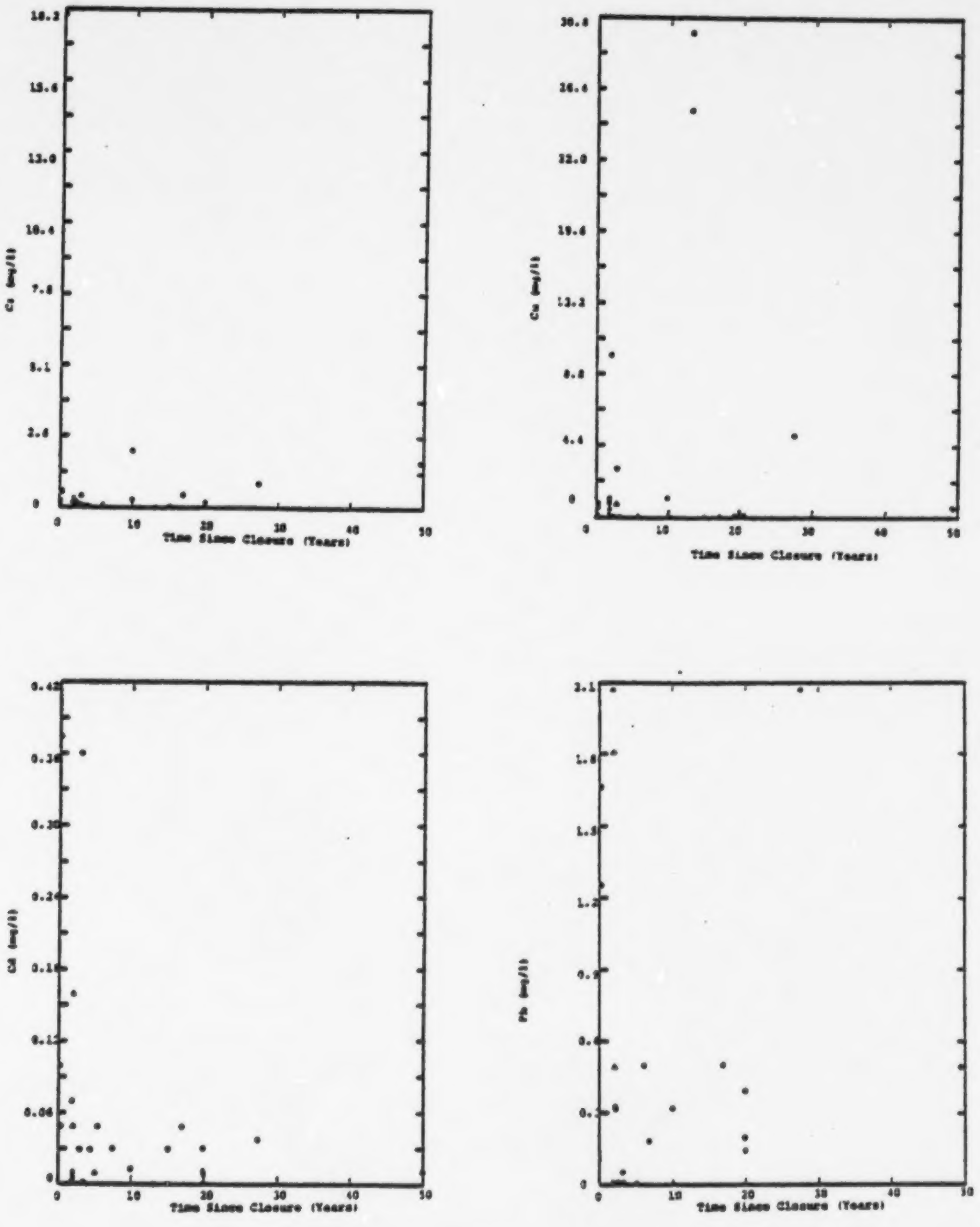

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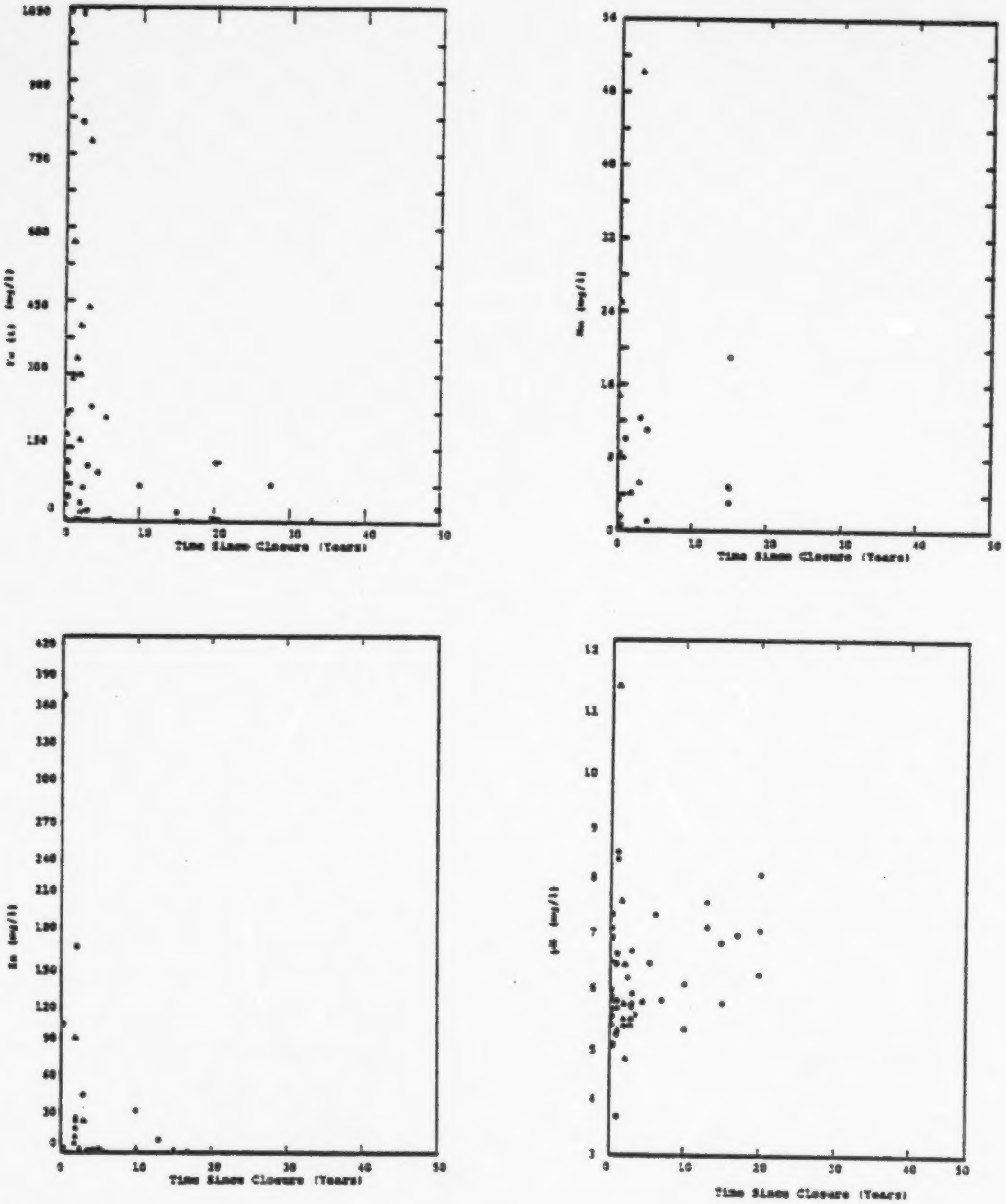

Figure 30. (Continued)

k = rate coefficient based on 10 (to be calculated)

Taking the logarithm of equation (34) we have

$$\log C_b = \log C_{b_o} - kt \quad \text{------------------------------(35)}$$

setting $\log C_b = Y$, $\log C_{b_o} = b$, $k = m$, and $t = X$, by substitution $Y = b + mX$, which is an equation of a straight line with slope m and Y intercept b. Thus, by plotting the logarithm of BOD_5 concentrations indicated by the boundary line shown in Figure 30 against the time value at those concentrations (semilogarithmic paper), the graph is a straight line with slope k, the rate coefficient. Because the graph is a straight line, the rate equation is of the form given in equation (34).

In most cases, the rate equation is adequate for the description of observed behavior in a limited time range. Therefore, equations derived are generally applicable for landfills of age greater than three years and less than thirty years. For some constituents, concentration data did not permit the formulation of a rate equation and rate coefficient, due to lack of sufficient data or the absence of an apparent trend. Trends were generally absent for heavy metal constituents. Apparently, the leaching of heavy metals does not necessarily follow the patterns established by the organic indicators, nutrients and major ions. It is speculated that the mass removal of heavy metals from refuse does not correspond to stabilization processes. Whereas the release of organic constituents (and probably nutrients and major ions) is closely tied to the various stages of refuse stabilization, heavy metals do not show this behavior. Apparently, considerable amounts of heavy metal pollutants are associated with inorganic refuse constituents, and their release may largely be a function of rate of flushing of waste and of the character of the leaching solution (i.e., a more acidic leaching solution may enhance metals release). The absence of trends may also be attributed to precipitation, dissolution, adsorption or complexation mechanisms which can retain or mobilize the metals within landfill microenvironments. It should be noted that considerably less data was available for heavy metals in leachates from older landfills (i.e., > 20 years), and that trends may appear as more data is gathered.

The leaching patterns of specific organic constituents and microbiological populations as a function of time since refuse placement require further elaboration. These topics are discussed below.

Organics--

Municipal landfill leachates often show the presence of organic compounds, which are either flushed out of the refuse by the leaching solution or produced during the biodegradation of refuse. Relationships between the general organic indicators (e.g., COD, BOD, and TOC) and landfill age (first page of Figure 30) indicate a substantial reduction in leachate organics following landfill closure. Studies have shown (Lin, 1966; Pohland, 1974) that free volatile fatty acids can represent the bulk of the organics present in leachates formed during the early steps of refuse stabilization, representing the acid fermentation stage of anaerobic degradation of the refuse. Figure 31 shows the trend of accumulation of organic acids produced by ground refuse in a test cylinder during early stages of stabilization (Lin, 1966). Figure 31 indicates that acetic acid was the only significant compound found in the first few days of leachate production, Thereafter, butyric acid production increased rapidly and amounts exceeded all other acids formed during the first two months of leachate production. Lin indicated a tendency for the amount of caproic acid to increase after 60 days with a prolonged period of fermentation.

With increasing landfill age, humic-carbohydrate-like compounds and fulvic-like materials become more predominate (Chian and DeWalle, 1977). Chian and DeWalle (1977) identified the various classes of organic compounds as a percentage of TOC and related variations to landfill age. Their results are presented in Figure 32. The volatile acids production, corresponding to the first stage of anaerobic degradation, represents the major organic fraction during the early years of landfill life. The volatile acid, carbohydrate, and protein concentrations decrease relatively rapidly with increasing landfill age. The aromatic hydroxyl compounds present in humic and fulvic-like fractions of leachate organics, show little decline with increasing landfill age, and are a major portion of identified TOC in older landfills.

The declining concentrations of specific organic classes of leachate TOC with increasing landfill age are also seen for organic degradation in natural environments. Carbohydrates degrade rapidly with increasing time of burial or increasing depth in sediment while a less rapid decrease is observed for amino acids (Degens, 1964; Ishiwatari, 1971). Aromatic compounds such as humic and fulvic acids, are generally less subject to decomposition than aliphatic compounds (Kohonova, 1961; Shapiro, 1964). Based on research findings and available literature (Chian, 1977) concluded that resistance of organics to bacterial degradation follows the pattern: free volatile fatty acids < carbohydrates, proteins, and humic-like substances < aromatic hydroxyl, carboxyl and fulvic-like substances.

The stability of organic compounds influences the leachability of trace metals. Many organic compounds containing nitrogen, oxygen, and sulfur can form soluble complexes with metals and thus promote leaching. Humic and fulvic acids are

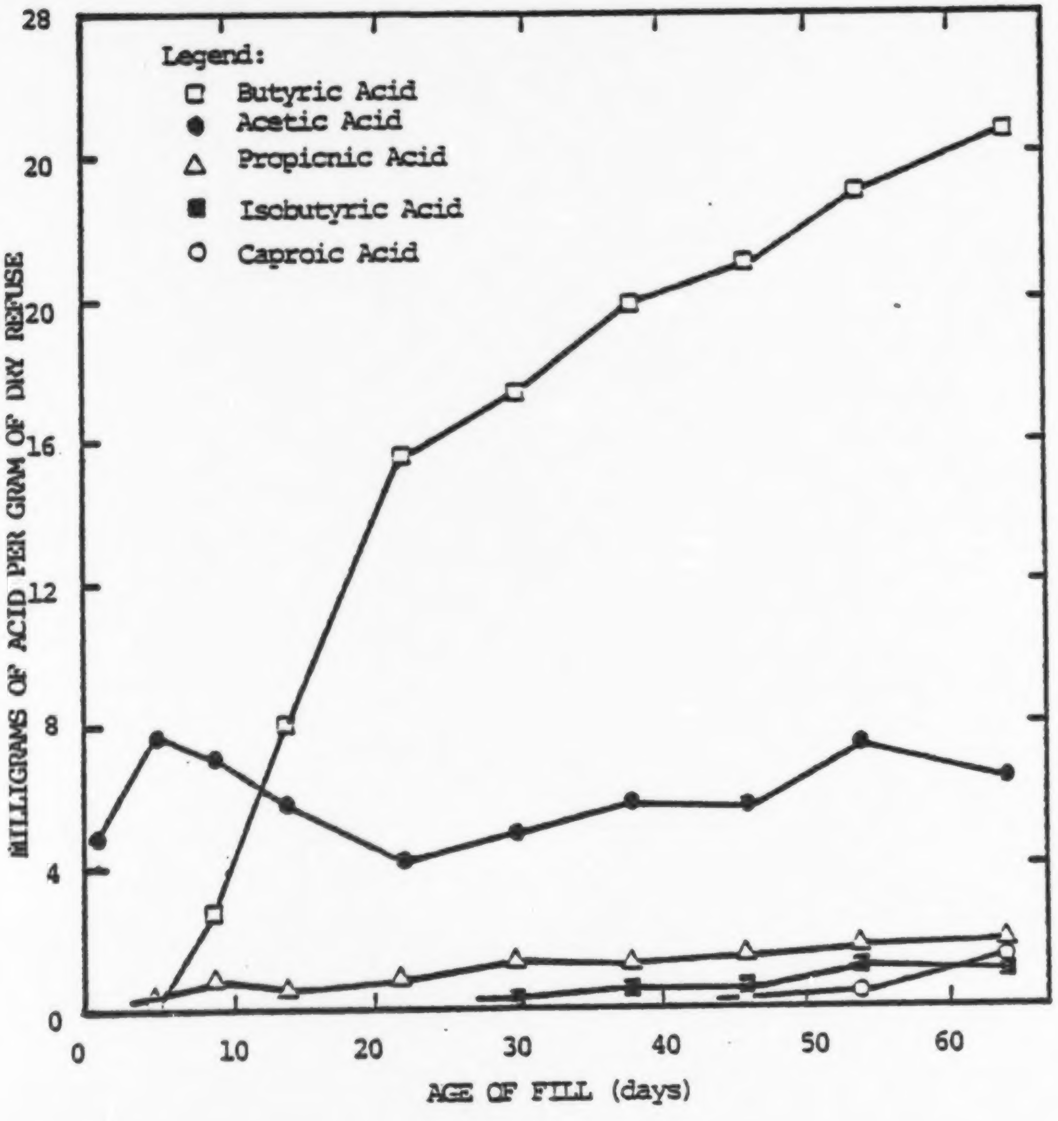

Figure 31. Volatile acids found in leachate during early stages of stabilization (Lin, 1966)

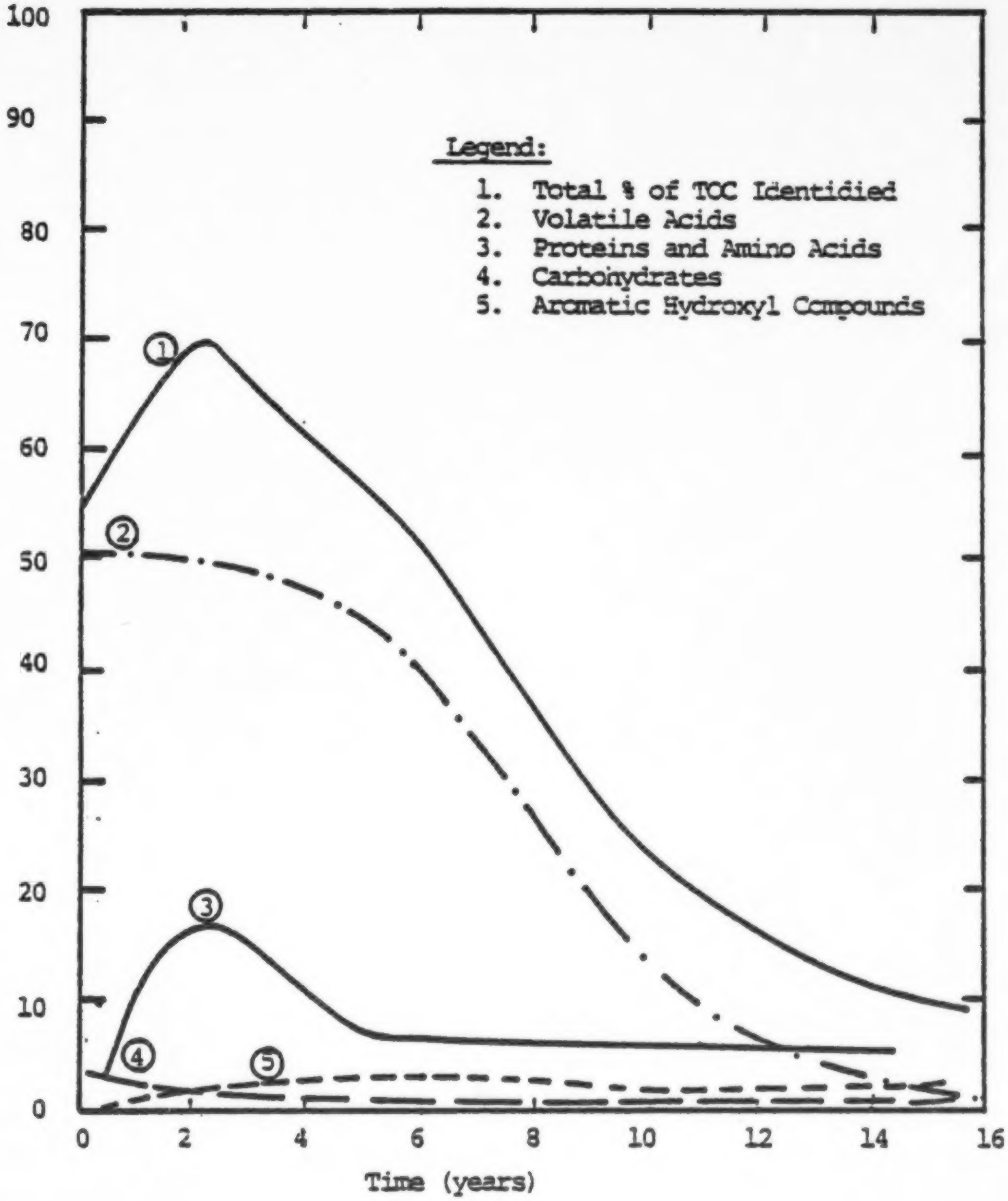

Figure 32. Trends in the identified fraction of leachate TOC vs. landfill age (Chian and DeWalle, 1977)

considered strong complexing ligands (Reuter and Perdue, 1977) and could be important in long term leaching of trace metals from a municipal landfill. In studying the complexation of cadmium by organic components of sanitary landfill leachates, Knox and Jones (1979) found that high (>10,000) molecular weight components contributed significantly to complexation. The authors suggested that these leachate components might contain phenolic hydroxyl groups having a stability constant toward cadmium of the order of 10^5. Conversely, Chian and DeWalle (1977) found that the majority of metals in leachates are not associated with organics having a molecular weight larger than 500. Knox and Jones (1979) stated that the role of complexation in determining the fate of metals in sanitary landfill leachate depends upon several factors whose importance had not yet been adequately investigated or reported in literature.

Microorganisms--

The effects of landfill age on the microbial populations of leachates was alluded to earlier in this section. In review, municipal solid waste leachates may contain significant bacterial populations, but these populations show a significant decrease in density with refuse age or time of leaching. A number of environmental factors are responsible for reducing these populations, including high landfill temperatures, low leachate pH, and the bacterio-static action of leachate organic acids and metals. There is some indication, however, that bacterial populations can survive in landfills and be present in leachates after extended periods, i.e., up to 9 years since refuse placement. More research is clearly needed in this area.

Enteric viruses are rarely found in landfill leachates. Because viruses are obligate parasites and cannot multiply outside host organisms, their numbers are also expected to decline with increasing landfill age. However, evidence suggests that viruses adsorbed to solid waste may be able to retain their infectious nature for extended periods. Viruses can be destroyed at the elevated landfill temperatures characteristic of the initial stages of refuse decomposition. The potential health hazard associated with viruses in leachate has yet to be resolved.

The presence of parasites in leachates has not been studied, though sewage sludge and fecal addition to municipal landfills provides a source for parasites to enter leachates. Parasitic cysts and ova can remain viable for extended periods in harsh environments, and their presence in municipal landfill leachates should be of public health concern. With regard to fungi in leachates,the paucity of research to date shows that the majority are non-pathogenic in nature. Their number in leachate appear substantially less than bacterial populations.

Rate of Water Application

The rate of water application to a landfill can influence its behavior in a number of ways, and concomittantly will affect leachate composition and pollutant removal. Assuming that the rate of water applied to a landfill is proportional to its infiltration to the fill material, a greater rate of water application will accelerate the attainment of landfill field capacity. Studies have shown that once field capacity is reached, leaching closely follows the pattern of moisture input, indicating that little change in internal landfill moisture occurs. Figure 33 shows the schedule of moisture addition and the observed leachate from Fungaroli (1975). Because of this "input-output" leaching behavior at field capacity, it is found higher rates of water application to a landfill will produce a more dilute leachate than lower rates of water application (Walsh and Kinman, 1981; Wigh and Brunner, 1981). Studies have also shown landfill microbial activity is related to moisture levels, i.e., biodegradative processes are diminished when landfill moisture remains below a certain level.* In light of these facts, it is feasible that high moisture levels in a refuse mass may result in leachate of distinctly different composition than leachates from refuse of lower moisture content.

Several studies have addressed the effect of water application rates on sanitary landfill leachate composition, the results of which are somewhat conflicting. Leckie, _et al_. (1979) constructed five large refuse test cells and subjected them to various moisture conditions through the controlled application of excess water, septic tank pumpings, and recycled leachate. A comparison of cells with and without continuous flow-through of water showed that continuous flushing brought about a significant trend toward lower concentrations of dissolved materials with time than the non-flushed cell. The results exemplify the important effects of dilution on leachate quality. Both inorganic and organic solutes were reduced at equivalent rates. The authors also note an accelerated stabilization of refuse materials during continuous flow-through. Using a laboratory lysimeter, 6 ft square by 13 ft deep, Fungaroli (1975) simulated seasonal variations in rainfall through artificial moisture applications. From his experiments, Fungaroli suggested that leachate produced during the slow attainment of a landfill's field capacity will probably exhibit initial pollutant concentrations different than a landfill in which substantial quantities are produced immediately. However, he adds, "once a system transients have eliminated, both systems should produce similar, but not necessarily identical, leachates." In another

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* Using laboratory-simulated landfills, Chian and DeWalle (1979) found that the moisture content of refuse must be above 75% but below 100%, based on dry weight, to maximize gas production.

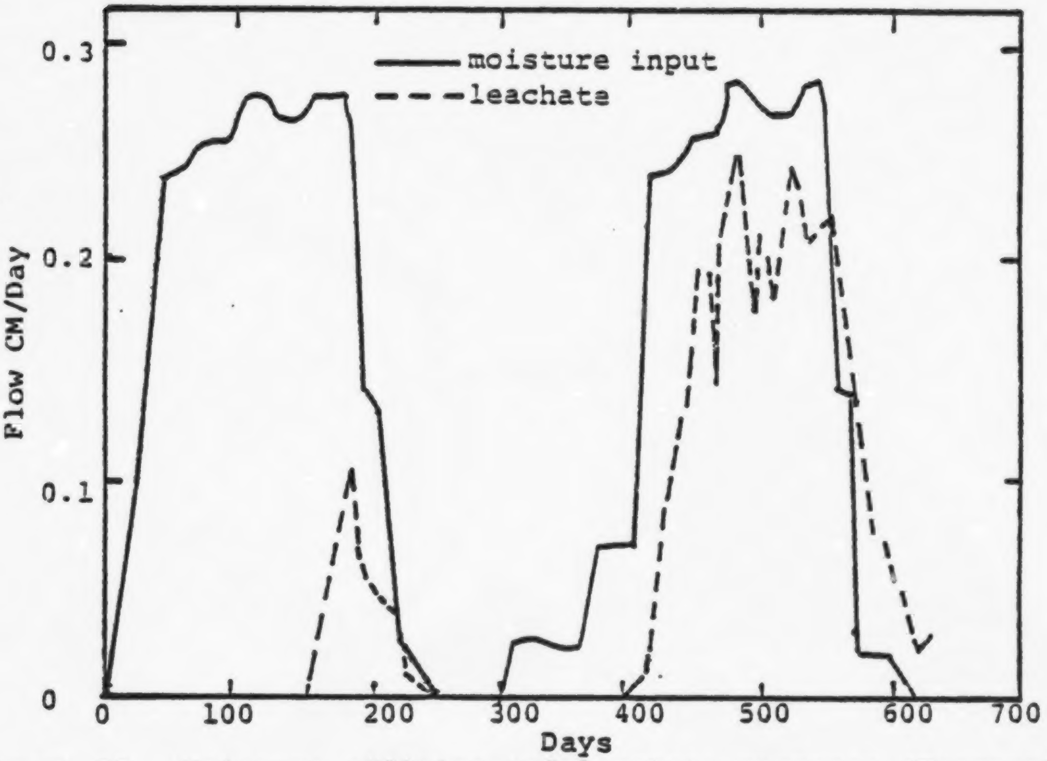


Figure 33. Moisture addition and leachate response (Fungaroli, 1971).

study using laboratory lysimeters with controlled moisture application, Revah and Avnimelech (1979) stated that contaminant concentrations in leachate were independent of water application rates. The authors demonstrated that for a given refuse type or density, the concentrations of leachate pollutants as a function of time were the same regardless of the volume of water passing through the column. These results contradict the results of virtually all other laboratory and field-scale investigations to date.

Two systematic efforts to determine the affect of moisture application on leachate quality were performed by Walsh and Kinman (1981) and Wigh and Brunner (1981) and provide perhaps, the best evidence of the influence of water application rates on leachate quality. In the Walsh and Kinman study, experimental lab-scale columns were constructed and subjected to a different amount of controlled moisture infiltration. The study continued for almost six years. The results showed that though constituent concentrations in leachates may vary with time depending on water application rates, the cumulative mass of constituent leached per cumulative leachate volume is generally the same, regardless of the infiltration rate. Wigh and Brunner (1981) found similar results. In their study, a pilot-scale cell and three landfill columns were operated for nearly eight years. Leachate contaminant concentration histories and mass removals were plotted as a function of cumulative leachate volume, and were found similar for varying water application rates. In comparing the similarities in the experimental results of several experimental landfills, Brunner (1979) demonstrated that meaningful comparisons of leachate quality can be made when results are expressed as a function of cumulative leachate volume. Figure 34 shows the results from Walsh and Kinman (1979) expressed in this manner. The horizontal axis normalizes the differences in infiltration rates over time. Although total solids on the vertical axis is a gross parameter which ignores the relative compositions of specific constituents, Figure 34 presents an ordered account of otherwise scattered leachate concentration data. This relationship between leachate constituent concentrations and cumulative leachate volume (or cumulative infiltration for landfills at field capacity) serves as the basis for several modelling efforts discussed later in this section.

Note that not all leachate constituents exhibit the abovementioned relationship between pollutant concentration and cumulative leachate volume. Heavy metal constituents, particularly iron, showed considerable variability in concentration as a function of cumulative leachate volume. Whereas the mass removal of other leachate constituents showed obvious decreasing trends as leaching continued, iron release appeared somewhat linear with time, even after 5 years (Walsh and Kinman, 1981). Apparently, iron is not as readily flushed out of a landfill as are other pollutants.

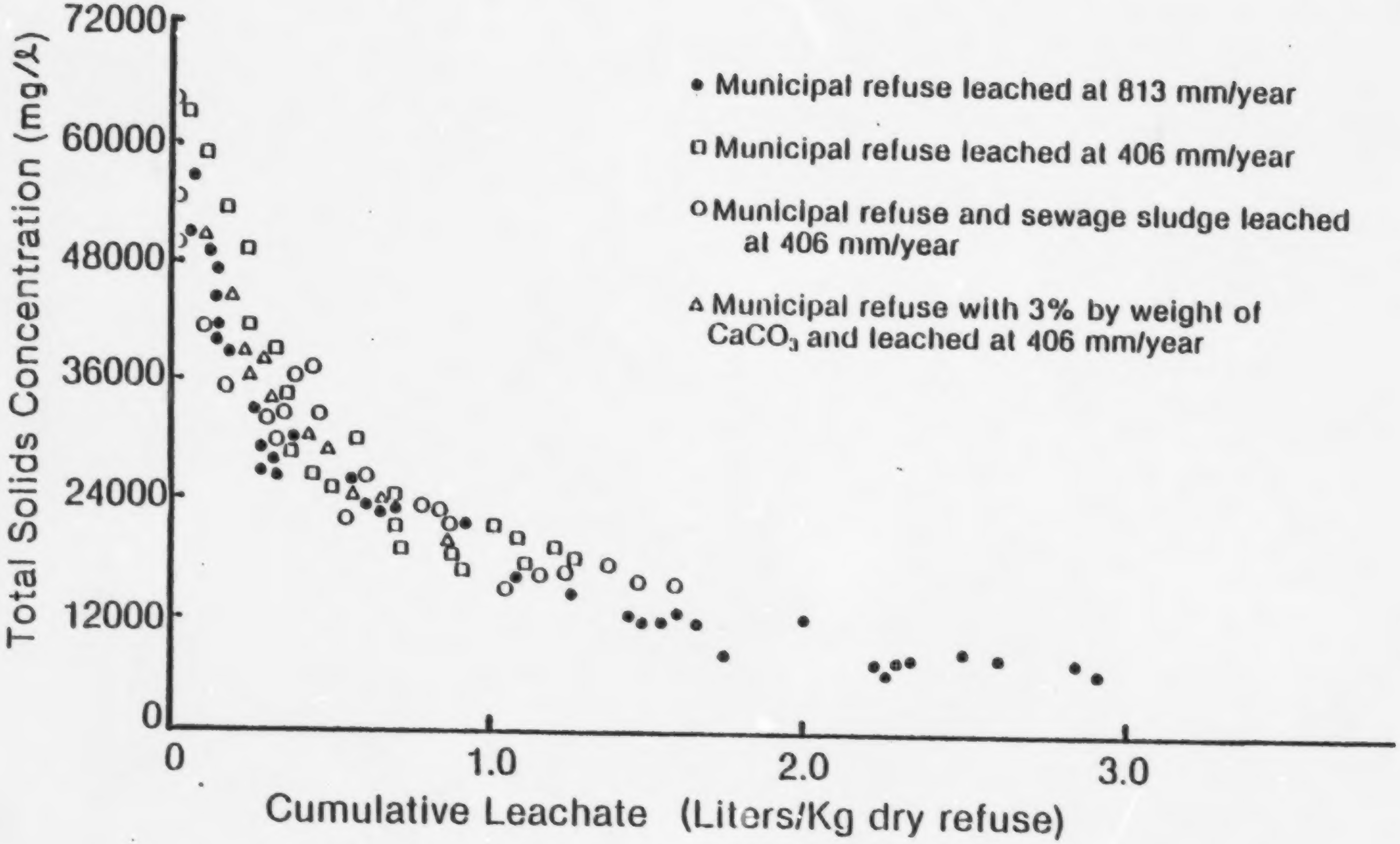


Figure 34. Leachate total solids concentration from experimental landfills subject to different moisture application rates (from Brunner, 1979)

Interestingly, organic leachate constituents appear to be flushed out of the fill as readily as most inorganic constituents. Straub and Lynch (1982b) suggest that high moisture flow rates can carry soluble organics and microbial cells out of the fill, and that in such cases microbial action appears to have a minor role in determining leachate quality. At lower flow rates, anaerobic microbial activity is thought to be a significant factor governing leachate organic strength, i.e., a significant reduction in organic material can be achieved.

In terms of landfill design and operation, these relationships reveal that the long term pollution potential of leachates can be diminished with initial high rates of moisture application. Provided that short-term leaching problems can be solved, high moisture application rates can remove the bulk of contaminants early in landfill life.

Depth of Leached Bed

Refuse depth, or lift height, is a design variable which operates in conjunction with the rate of water application to influence leachate composition. To illustrate, consider a solid waste landfill consisting of compacted refuse. Some fraction of the volume of the landfill will be void spaces capable of holding water or gas. Water entering the top surface of the landfill and travelling down through the refuse will move from one void space to the next until it eventually reaches the bottom surface of the landfill. Polluting material coming in contact with the percolating water is transferred from the solid to the liquid phase due to a concentration difference between them. Thus, water percolating through refuse will successively accumulate contaminants until the solubility limit of the leaching solution is attained and no further exchange can occur. For a deeper landfill, the chances of leachate attaining its solubility limit for various constituents is greater than for a shallow landfill, because of greater contact time between refuse and percolate.

However, much depends on the rate of water percolating through the refuse. Water percolating slowly through a landfill provides greater contact time between the solid and liquid phases, and increases the chances for a high strength leachate to result. For high infiltration rates, contact time is diminished between the two phases and a more dilute leachate is expected. The concept of a residence time in the refuse for a given volume of liquid can be used to anticipate the effect of changes in both infiltration rates and refuse depth. That is, for any liquid travelling the full depth of the refuse, the residence time (and leachate strength) should increase directly with the depth and inversely with the infiltration rate.

One of the earlier studies confirming the importance of refuse depth was performed by Qasim and Burchinal (1970). These researchers found substantially greater constituent concentrations in leachates from taller concrete cylinders (exceptions were hardness and calcium). Deeper fills under similar conditions of precipitation and percolation were found to require more water to reach saturation, require a longer time for decomposition, and distribute the bulk of extracted material over a longer period of time. Laboratory column studies performed by Phelps (n.d.), demonstrated the effects of depth on leachate composition. Columns filled with refuse of varying depths (i.e., 38, 76, and 114 cm) and subjected to equal annual net infiltration showed distinctly different concentration histories over a 560 day period. Figure 35 illustrates the concentration histories of BOD and chloride, as well as indicate the simulated concentration histories from the model developed in conjunction with the study. The figure clearly shows that higher concentrations of leachate contaminants emanated from the deeper columns. The deeper columns allowed greater residence time in the refuse for a given volume of percolate.

Given that saturation limits of leachate might be reached for a deeper landfill, and that increased infiltration rates allow for a greater mass of contaminants to enter the leachate, pollutant concentrations in leachate should decrease more rapidly in time for shorter residence time or greater infiltration rates. In other words, since the ratio of the mass of liquid per unit time and the mass of refuse it passes through is greater at shorter residence time, polluting material is flushed out of the refuse more quickly at greater infiltration rates.

Landfill Temperature

Although chemical and biological processes generally are temperature-dependent both in terms of rate and ultimate equilibria, at the present time it is not reasonable to speculate on the effect of temperature on leachate composition and refuse stabilization except in general terms. Research has shown significant differences in landfill (experimental and field scale) temperatures with time, depth, and mode of operation. Major studies and results are summarized below.

Temperatures in the refuse mass are found to reach a maximum generally within a few days of refuse placement. The high temperatures correspond to the aerobic decomposition phase of refuse stabilization (Chian, et al., 1977). Merz (1954) found maximum temperatures in a refuse bed to range from 138° to 157°F within 8 days following refuse placement in wooden percolation bins. Fungaroli and Steiner (1979) observed a peak temperature of 150°F at the center of a laboratory lysimeter to occur within 10 days of refuse placement. At depth, Wigh (1979) observed a peak temperature of 110°F to occur the day of lift construction,

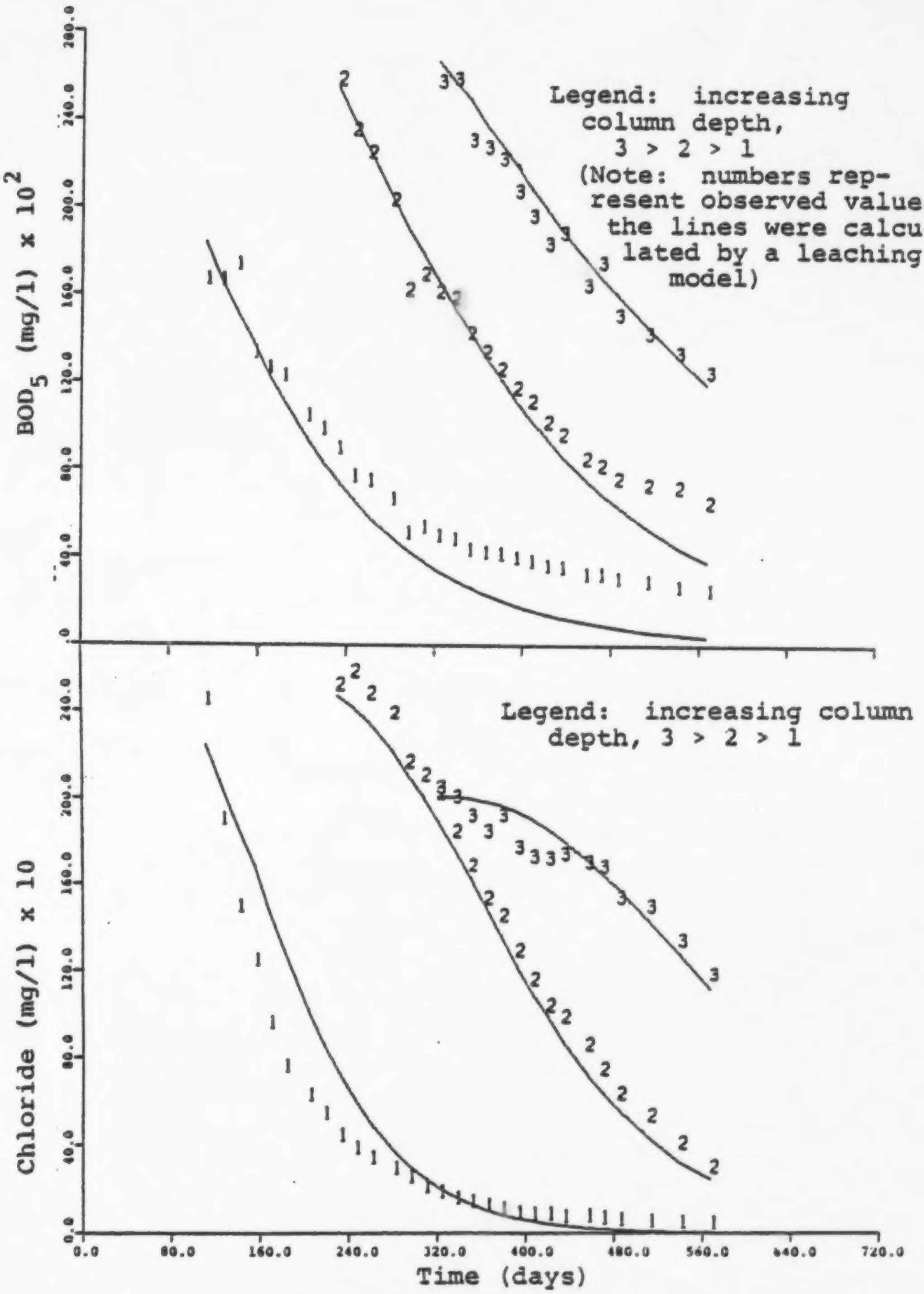


Figure 35. Leachate concentration nistory curves for simulated landfills of different depths (from Phelps, n.d.)

indicating a short period of active aerobic decomposition.

Following the aerobic decomposition phase of refuse stabilization temperatures at depth gradually decrease, signalling the onset of anaerobic decomposition. Fungaroli and Steiner (1979) found lysimeter temperatures to stabilize at approximately 80°F in 60 days following refuse placement. Evidence from several studies, however, indicates that temperatures within the refuse mass are not static, rather there is a tendency for refuse temperatures to vary with ambient air temperatures, particularly with seasonal variations. Leckie, et al. (1979), for example, studied temperature variation in compacted refuse of five field scale experimental cells. In general, there was an apparent thermal response of the upper several feet of landfill to the mean ambient temperature, which varied seasonally in a range of approximately 20°C over the annual cycle. Deeper refuse material (e.g., 3 meters) tended to show a much smaller thermal response. The authors noted that the effect of the variation in mean ambient air temperature was significant only to the extent of the variation over an annual seasonal cycle. Temperatures in the refuse beds followed the ambient temperature cycle with an observed time lag in their response.

Other studies noted similar landfill thermal response to ambient air temperatures. The refuse mass in test cells studied by Wigh (1979) showed response to seasonal temperature variations, though the temperature amplitudes within the cells were reduced with increasing depth. Temperatures at 2 m depth ranged between 47 and 67°F over the year. Merz (1954) found refuse temperatures in bins to fluctuate with seasonal temperature changes, from a low of 75°F in February to a high of near 100°F in summer months.

The thermal response of refuse beds to changes in ambient air temperatures correspond to those presented in literature on both the diurnal and annual cycles of heating and cooling of soil, which also shows a typical time lag in heating and cooling at depth compared to the annual or diurnal surface cycle (Gieger, 1965; Singer and Brown, 1956). From the information presented above, it is expected that there is a landfill depth below which temperatures would remain essentially unchanged year round, and this temperature is typically close to the average annual air temperature near the ground surface. Because of active anaerobic decomposition, landfill temperatures are expected to exceed soil temperatures at a given depth (Riehl, 1972; Strahler, 1971).

Elevated landfill temperatures may exhibit significant adverse effects on the survival of pathogenic agents. Englebrecht and Amirhor (1975) studied the stability of enteric bacteria to elevated temperatures by adding cells to leachate at two different temperatures (72°F and 131°F). The persistence of enteric bacteria was found to be significantly less at 131°F. Wigh (1979) inoculated shredded refuse with poliovirus type 1 and *Salmonella derby* and placed it in a field test cell to evaluate the microorganism movement. Both pathogens were found to be inactivated in less than 10 days. Temperature appeared to be the essential organism inactivation factor. Estimated peak temperatures were approximately 140°F. From these observations, elevated temperatures corresponding to the aerobic phase of refuse decomposition appear to be unfavorable for pathogen transmission via leaching though additional research is required for verification. Further assessment should be conducted.

Summary

Research on several factors which influence the composition of municipal landfill leachates has been reviewed. Attempts were made to identify how some of these factors can be controlled in order to modify the character of leachate. In summary, refuse processing, including shredding and baling, can significantly affect leachate composition. Refuse shredding tends to increase landfill field capacity, can increase the concentration of pollutants in leachate in comparison to unshredded refuse leachates, and leads to a increased rate of pollutant removal per volume of leachate in comparison to unshredded refuse. Conversely, baling promotes the production of a more dilute leachate, a delayed attainment of landfill field capacity, and smaller mass removal per volume of leachate compared to shredded or unbaled refuse. Regarding the rate of water application, increased rates can lead to a more dilute leachate and greater mass removals as a function of time than lower rates of applied water. High moisture application rates can remove the bulk of refuse contaminants early in landfill life. Increasing landfill depth will generally promote a higher strength leachate, though at high infiltration rates the effect is diminished. Landfill temperature, a largely uncontrollable factor influencing leachate composition, have been shown to fluctuate with seasonal ambient temperature variation near the landfill surface, but amplitudes are less pronounced with increasing landfill depth.

EFFECTS OF ADDING SLUDGES TO MUNICIPAL LANDFILLS

The addition of municipal wastewater treatment plant (WWTP) sludges or industrial sludges to municipal landfills is reported to provide both beneficial and adverse effects on leachates. In various studies, WWTP sludge and refuse admixtures have accelerated the rate of stabilization of biodegradable organic matter. However, sewage sludge/refuse codisposal alternatives may pose greater adverse environmental and public health impacts than conventional refuse disposal. Concern has arisen that the addition of industrial waste may result in the occurrence of various toxic elements in leachates in excess of that encountered in municipal refuse leachates. The addition of industrial sludges may also pose a deleterious impact on the biochemical stabilization processes within the landfill. These effects of adding sewage sludges and industrial sludges to a municipal landfill are identified in the following sections.

Addition of WWTP Sludges

A number of column and field studies have been performed to investigate the effects of sewage sludge codisposal on leachate composition. Pohland (1975) found that seeding municipal refuse with primary sewage sludge in test columns increased rates of biological stabilization of organic pollutants in refuse and leachate in comparison with non-seeded columns. Emcon Associates (1974) reported that the addition of septic tank pumpings to municipal refuse accelerated the methanogenic process, as evidenced by increased methanogenic activity related to the control cell (refuse only). The study also showed that leachate bacterial loading did not exceed the refuse-only cell; leachate organics and TDS were reduced, while inorganic ion concentrations were largely unaffected. Stone (1974) found that admixed WWTP sludge and refuse produced an acidic leachate with a higher BOD_5 than the solid waste leachate, but that the chemical composition did not significantly differ. Lu, et al. (1982) determined that codisposal leachates posed no more of a pollution threat than municipal refuse leachate, though codisposal can accelerate leachate formation through the additional moisture provided by the sludge.

Leachates of WWTP sludge and refuse codisposal operations appear similar to conventional refuse leachates. Reported investigations differ in regard to whether sludge can enhance removal of organic matter from refuse and leachate through accelerated stabilization. A paucity of information exists regarding the potential public health impacts of sewage sludge/refuse codisposal landfilling (Lu, et al., 1982). The degree of health risk associated with WWTP residuals is proportional to the degree of microbial survival from the sewage treatment processes. An increase in pH from lime conditioning, heat treatment, chlorination, dewatering, and incineration will significantly diminish the survival of pathogens in sludge.

The transmission of pathogens and toxic material from WWTP sludge/refuse landfill operations can occur primarily via surface waters and ground waters. Epidemiological evidence of disease occurring as a result of WWTP sludge codisposal landfilling is lacking. Nitrate-nitrogen and enteric pathogens (i.e., bacteria and viruses) appear to represent the greatest potential to ground-water contamination from leachates of sewage sludge/refuse codisposal operations.

Addition of Industrial Sludges

Industrial wastes vary greatly in content, physical characteristics, and potential for environmental degradation. In many cases, the potential for inclusion of highly toxic contaminants in leachates is greater than for WWTP residuals or municipal wastes. Limited research has been conducted concerning the potential toxicity of various industrial sludges. The few studies on the subject, however, are summarized as follows.

Streng, et al. (1976) evaluated the leaching characteristics of several industrial residuals admixed with municipal refuse, including: refinery sludge (RS), battery production waste (BPW), an electroplating waste sludge, (EW), an inorganic pigment sludge (IPW), a chlorine production brine sludge (CPBS), and a solvent based point sludge (SBPS). Notable components of these sludges were: Cu, Fe, Hg and moisture for RS; Cd, Fe, Cu, Pb and asbestos for SBPS: Sn, Sb and moisture for BPW; Cr, Fe, As, Cd, cyanide, and moisture for EW; Be, Cl, asbestos and moisture for IPW: and Ni, Pb, Cl, asbestos, and Hg for CPBS. Sustained maximum concentrations of leachate contaminants as compared to refuse are presented in Table 41. Leachate contaminant concentrations from a sludge and refuse mixture exceed those of solid waste, particularly for those components in abundance in the respective sludge. Other important observations included: (1) field capacity was obtained earlier for the majority of the sludge/refuse cells, probably due to the added sludge moisture, (2) industrial sludge leachates exhibit both a viricidal and bacteriacidal effect.

The leaching trends of chemically stabilized and unstabilized industrial wastes in conjunction with municipal solid waste (MSW) was investigated by Myers, et al. (1979). The results are summarized as follows:

(1) When codisposed with MSW, the release of major metal contaminants from treated or untreated electroplating sludge was not observed.

(2) Untreated chlorine production brine sludge when codisposed with MSW, releases significant quantities of Al, Cd, Cu, Cl, Hg, Na and other dissolved solids. Chemically stabilized chlorine production brine sludge when codisposed with MSW significantly reduces the mass release of toxic metals (e.g., Cd, Hg, and Cr).

(3) Codisposal of MSW with calcium fluoride/sewage sludge apparently improved leachate quality with respect to BOD, COD, TOC, alkalinity, pH, and iron.

TABLE 41. SUSTAINED MAXIMUM CONCENTRATIONS OF LEACHATE CONTAMINANTS FROM STRENG (1976)

Waste Cell	SW[b] 4	RS[c] 9	BPW[d] 10	EW[e] 12	IPW[f] 13	CPBS[g] 14
Vanadium[h]	<0.03	<0.03	([0.29])[ij]	<0.03	([0.23])	<0.03
Arsenic[k]	12.2	<3.0 [ℓ]	([30.0])	<3.0	13.0	([54.0])
Selenium[k]	<5	(13)	<5	([230])	([10.0])	<5
Antimony[k]	<5	<5	<5	<5	([17])	<5
Tin[k]	150	(800)	(1200)	(3000)	(2400)	([8000])
Sulfur[h]	674	NSA[m]	491	68	(846)	([3362])
Asbestos[n]	<100	<100	<100	<100	([6000])	([3000])
Cyanide[k]	<1	<1	([1.2])	<1	5.0	<1
Aluminum[h]	<0.05	(1.2)	(8.1)	<0.05	(5.4)	([11.2])
Phenol[h]	4.3	([22.9])	NA[p]	NA	NA	NA
Beryllium[k]	<0.1	(0.9)	([45.0])	([30.0])	4.5	([48.0])
Lead[k]	1510	1450	([3380])	(1820)	(2140)	([6460])
Titanium[h]	<0.1	<0.1	<0.1	<0.1	<0.1	(0.6)
Mercury[k]	19.7	(31.2)	7.2	(26.9)	(67.1)	([328])
Boron[h]	18.1	2.8	13.3	(22.0)	(24.0)	([83.0])
Nickel[k]	1360	1010	1319	([3505])	(1490)	([6020])
Zinc[h]	4.3	2.1	(5.2)	0.5	([11.2])	(8.0)
Summation		4 (6) [1]	3 (8) [5]	3 (7) [3]	0 (13) [5]	0 (12) [10]

[a] Higher concentrations may have been detected.

[b] Solid Waste.

[c] Refinery Sludge.

[d] Battery Production Waste.

[e] Electroplating Waste.

[f] Inorganic Pigment Waste.

[g] Chlorine Production Brine Sludge.

[h] mg/ℓ

[i] ( ) indicates exceeded solid waste.

[j] [ ] indicates exceeded maximums.

[k] μg/ℓ

[ℓ] Underscore indicates less than solid waste.

[m] Insufficient sample available.

[n] Fibers/ℓ

[p] Not analyzed.

Utilizing residential-type solid wastes alone and in conjunction with a metal plating sludge, Pohland and Gould (1980) indicated the possibility for significant reductions in leachate heavy metal concentrations and key roles of redox potential, pH and sulfides, hydroxides and hydroxy-carbonates in sequestering metals. Studies designed to determine the limits of heavy metal loadings which can be sustained during codisposal without deleterious effects on landfill stabilization processes are required. Studies on the reduction of these effects by addition of appropriate sequestering agents are also needed.

There is still little known concerning the effects of industrial sludge on refuse decomposition processes and leachate produced during decomposition. Evidence suggests, however, that the addition of sludges, particularly those high in trace and heavy metals, will result in elevated metal concentrations in leachates. There is also evidence suggesting certain wastes or landfill conditions can immobilize trace and heavy metals.

LEACHATE COMPOSITION MODELS

The development of a descriptive and predictive model of the leaching of municipal landfills and the composition of leachate can be advantageous for the following reasons (Phelps, n.d.):

1) To assess the potential impacts of leachate on receiving waters;

2) To aid in the design of a leachate treatment system;

3) To help estimate the concentration of contaminants entering an underlying ground-water system so as to determine leachate concentration in that system;

4) To assess the possible effects of codisposal of various liquid and/or semi-solid wastes with municipal solid waste in landfills.

This section of the report has thus far described leaching processes and significant factors affecting the generation and composition of municipal landfill leachates. Though the leaching process involves a number of variables whose interactions are poorly understood, some generalizations can be made. For example, little leachate is collected at the bottom of a landfill until the field capacity of the refuse has been reached. The field capacity of refuse has been similar among various studies, and appears to be a function of in-place density (Fungaroli, 1971; Revah and Avnimelech, 1929). Once the landfill has attained field capacity, leaching patterns closely follow the input of moisture to the landfill, indicating that little change in internal moisture occurs. In addition, though the concentration of contaminants in leachate may vary from study to study, there is a strong correlation between leachate concentration and the total volume of leachate produced (Wigh, 1979; Walsh and Kinman, 1981). Leachate contaminant concentration reach a maximum level near the onset of leaching and decrease as leaching continues.

Whereas models developed for estimating leachate generation (Chapter 3) rely on observations of field capacity, water input to the landfill, and a water mass balance to route moisture through a landfill, they make no attempt to predict leachate quality. In this section, modeling efforts which describe and predict the quality of leachates from municipal landfills are reviewed. The basic leaching behavior generalizations described above as a starting point, the models attempt to estimate leachate quality as a function of time or cumulative leachate generation.

Curve Fitting Approaches

There are generally two approaches used when attempting to model the composition of leachates as a function of time or cumulative leachate volume. One approach is to quantitatively describe the physical, chemical, and biological processes which occur (or are assumed to occur) during leaching. Because landfills are dynamic environments, these types of models tend to be complex, and make a number of simplifying assumptions. The other approach is to avoid mathematical expression of leaching mechanisms and to focus merely on leachate concentration histories. In this approach, empirically equations are developed which describe the shape of curves derived from experimental data on the concentration of leachate constituents versus time or cumulative leachate volume. Because the concentration histories of leachate constituents are highly dependent on variables such as refuse character, refuse density, landfill depth, and rate of water application, these "curve-fitting" models are generally effective only in modeling the experimental data from which they were derived.

Revah and Avnimelech (1979) used the curve-fitting approach to model the concentration history of organic carbon, volatile acids, TKN, NH_3, NO_3, Fe, and Mn in leachates from refuse-filled columns. The steeply declining concentration of these leachate constituents was described through an exponential function:

$$C = ab^t \text{-----------------------------------} (36)$$

where:
C = the concentration of leachate constituent,
t = time,
a = concentration at t = o, and
b = the exponential base

Equation (36) above was used by the authors to determine the length of time (t_f) needed for a leachate constituent to reach a certain limiting concentration (C_f), i.e., the concentration below which the leachate is not endangering water quality. The model parameter t_f was calculated for columns containing raw and shredded refuse, varying refuse densities, and for columns filled during different filling seasons. According to the calculation, potential pollution by sanitary landfill leachates, continues for about 3 to 15 years after the placement of the solid waste. The fastest decay was associated with the least dense columns, filled during the winter. The leaching period is extended for shredded and compacted refuse. These observations are in agreement with previously reported research on the effects of refuse shredding and compaction. The authors also report a general similarity between field data and those obtained in this model system, indicating that exponential decay adequately characterizes the leaching process.

Wigh (1979) also employed the curve-fitting method to model leaching patterns. Noting the relationship between leachate concentration and the total volume of leachate produced, Wigh proposed an equation for two consecutive first-order reactions having the form:

$$C = \frac{k_1 C_i}{k_2 - k_1} \left(e^{-k_1 v} - e^{-k_2 v}\right) \qquad \text{----------------} (37)$$

where:
C = the concentration at any volume of leachate
k_1 and k_2 = rate constants
c_i = unknown concentration related to the contaminant mass available in the refuse
v = the volume of leachate collected per unit of surface area.

Since volume of leachate collected is some function of time, the equation could be changed to a time dependent function.

From experimentally determined leachate concentration histories, a trial and error process was used to determine the best visual fit for reaction curves and rate constants. Figure 36 shows a reasonably good fit for leachate COD and chloride. While the equation can fit well the concentration histories of various leachate constituents in this experiment, the fit is an empirical one, with constants having significance only for the particular experimental landfill for which they were evaluated.

Knowing the rate constants, it was possible to compute C_i in equation (37) and solve for the total mass of a refuse constituent that might be leached. By integrating equation (37) through an infinite volume of leachate, the resultant total leachable mass in mg per unit of surface area was:

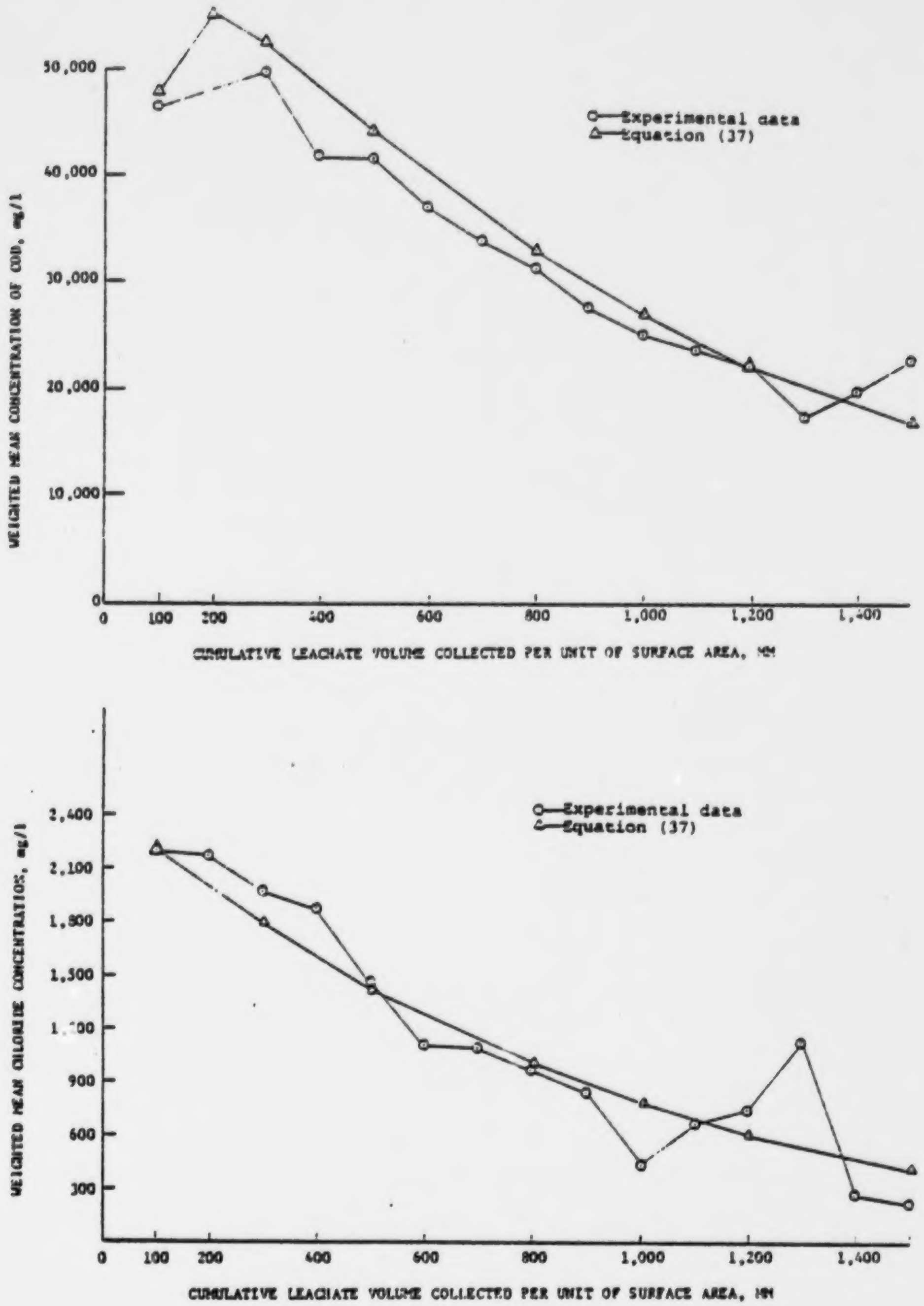


Figure 36. Simulated and experimentally derived concentration histories for COD and chloride (from Wigh, 1979)

$$M = C_{i/k_2} \text{-----------------------------------} (38)$$

The total mass of dry refuse available for leaching and percentage removal at 1500 mm of leachate is presented in Table 42. If equation (38) is an accurate representation of the leaching process, extraordinary removal had occurred after only 1500 mm of leachate.

TABLE 42. TOTAL AVAILABLE MASS REMOVAL (from Wigh, 1979)

Parameter	Available Total Mass[a]	Removal at 1500 mm[b]
COD	89.6	70
Sulfate	1.41	75
Chloride	2.73	81
Magnesium	.500	86
Iron	3.26	31

[a] g/Kg of dry refuse

[b] percent removed obtained by dividing mass actually removed at 1500 mm by total mass calculated from Equation 38.

Process Modeling Approaches

There has also been a few attempts at developing equations which represent a fundamental description of the leaching process which can be applied to other landfills. These models help describe and predict moisture and contaminant movement in sanitary landfills. The models are sensitive to such factors as refuse placement and composition, hydraulic phenomenon and landfill configuration. The models range in levels of sophistication-the more sophisticated the model, the more responsive the models must be to landfill construction and solid waste or media properties.

Qasim and Burchinal (1970) assumed a simple physical process to describe leaching from sanitary landfills. The theory of column operation was applied to the leaching of chloride during the vertically downward movement of water. Chloride was used as a surrogate parameter of landfill leachate. The approach assumed that the chloride concentration may be determined on the basis that adsorption occurs at a rate proportional to the concentration of solute in the liquid phase, and by the difference between the actual and maximum possible concentration of the solute adsorbed on the solid particle. The model is responsive to the physical parameters of depth of refuse, refuse compaction, and cumulative leachate volume. Experimental and theoretical concentrations of leachate constituents in leachate samples showed fair agreement, with maximum deviations of 30%. Theoretical calculations tended to underestimate chloride concentrations as leaching continued. The duration of the leaching experiments was only 163 days.

Another physical process model of the leaching of solid waste has been developed and tested by Phelps (n.d.). In concept, the model likens a landfill to a packed column, wherein a solutoin percolating through the column picks up contaminants from the solid phase due to a concentration difference across a stagnant film of liquid on the particles in the solid phase. The concentration on the particle surface is assumed to be proportional to this amount of leachable material per unit volume of refuse (voids and liquid included). The mathematical derivation of the model is quite complex, involving a series of partial differential equations.

The model was tested for a special case assuming a constant rate of infiltration and low initial refuse moisture content. The analytic expression is a function of time, refuse depth and refuse moisture content, and thus requires only three empirically derived parameters. The experiments used to test the model cover a six-fold range in both refuse dpeth and infiltration rate. The experiments also cover two sizes of refuse particles and the presence or absence of septic tank sludge. Six test results for each experiment were used: BOD, total residue, calcium, iron, COD, and chloride. The correlation coefficient, expressed as R^2, between the calculated and observed curves for concentration vs. time was calculated. The resulting values of R^2, generally exceeded 0.9.

The calculated curves generally agree with the data within a few percent over the initial upper 75% to 88% of the concentration values. Figure 35 shows the calculated and observed concentration curves for BOD_5 and chloride for a given set of experimental parameters. The figure shows good agreement from the time leachate first appears to about twice that time. For longer times, the observed concentration values are consistently somewhat above those calculated. The extent to which the later data values exceed those calculated was found to depend on infiltration rate and the addition of septic tank sludge.

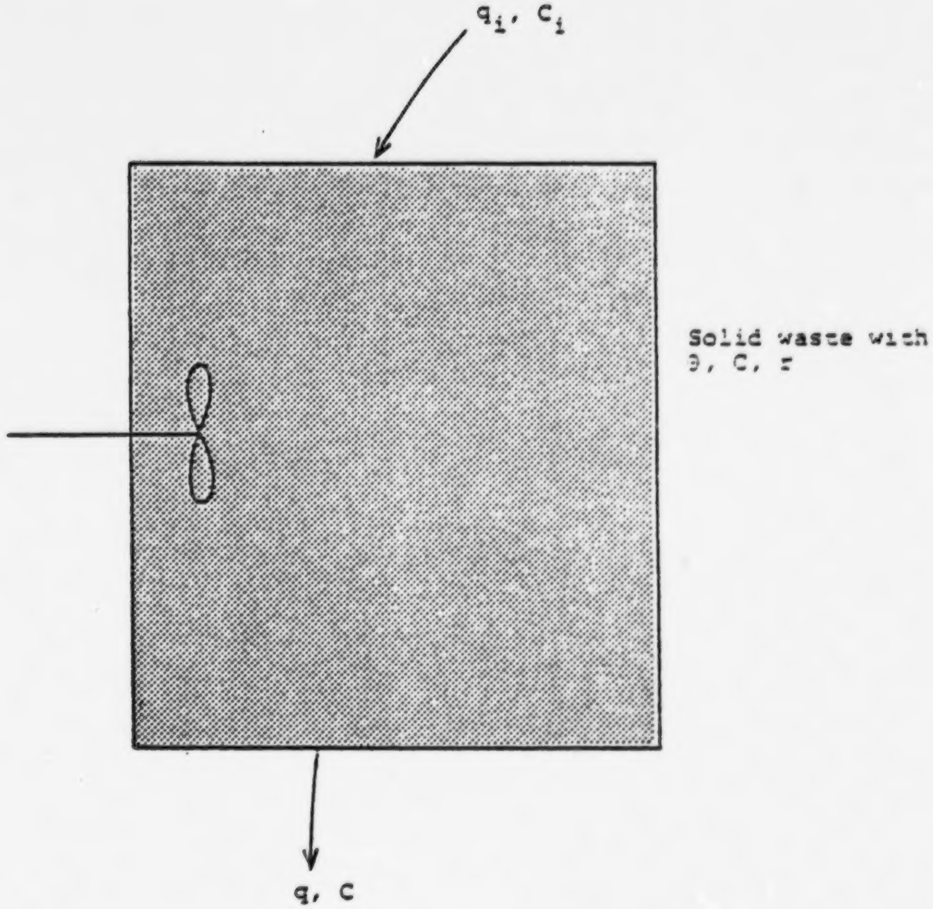
The trends in the empirical parameters of the Phelps study indicate the effects of the various experimental operating conditions: infiltration rate, particle size and septic tank sludge addition. These empirical parameters provide a basis for direct comparison of leaching curves for widely varying conditions. However, it is questioned whether the observed trends in the empirical parameters provide a basis for applying the model in its present form to field situations.

Finally, a series of models have been developed by Straub and Lynch (1982a) which describe separately the leaching processes of inorganic and organic constituents in landfills. These models represent the best attempt to systematically describe and predict leachate volume and quality from sanitary landfills. Like other models discussed thus far, the fundamental relationships among landfill variables provide a framework for the modeling. Unlike the previous models, however, the level of sophistication has been greatly increased by the formulation of specific hydraulic, physical/chemical, and microbial processes. The types of leachate behavior simulated with the models include:

1) Hydraulic behavior. This model simulates the moisture flow in a sanitary landfill based on equations of unsaturated flow in a porous media. The model is an extention of Darcy's equation, insofar as a finite difference numerical solution is developed. While the water balance method of modeling leachate production (Section 3) may yield reasonable predictions of leachate production in some cases, it does not describe internal moisture flow details or contaminant movement. This model provides a continuous description of moisture flow, which can subsequently be used in modeling of contaminant transport. The model applied knowledge of the hydraulic properties of refuse and soil layers, as well as properties of soil layers which affect landfill moisture flow. Typical forms of hydraulic functions were applied to refuse and yielded good simulation of moisture movement. However, specific empirical information on refuse hydraulic characteristics is limited, requiring additional research.

2) Inorganic leachate composition. The concentration and relative strength of inorganic contaminants (represented by total solids) is described by both a simple single reactor model and vertically distributed unsaturated flow model. The simple reactor model considers the volume of refuse as a single well mixed reactor (Figure 37), and that leaching can be described as a simple exponential function of cumulative leachate volume. The leaching pattern of inorganic constituents is described as a function of the net infiltration rate of moisture, the volume and field capacity of the refuse, and the rate of generation of dissolved substances.

The vertically distributed model describes contaminant transport in an unsaturated porous medium, and was developed


q_i, C_i
Solid waste with
θ, C, r
q, C

q_i = inflow rate, vol/time
C_i = inflow concentration, mass/vol
q = outflow (leachate) rate
C = outflow concentration
θ = volumetric moisture content
r = contaminant generate rate, mass/time/vol

Figure 37. Single well-mixed reactor model of a landfill (Straub and Lynch, 1979)

to be used with the unsaturated moisture flow model as described in (1) above. This one-dimensional kinetic model allows the simulation of the continuous transport of contaminants throughout the vertical depth of the landfill, for inputs of moisture content, contaminant concentration, and arbitrary source terms. The dilution, dispersion, and convection of these contaminants by the flow of water is described. The mathematical derivation of each of these models is extremely complex, and interested readers are asked to consult Straub (1980) or Straub and Lynch (1982). Figure 38 shows the observed total solids concentration versus time for an experimental test cell as well as results simulated by the vertically distributed model.

In general, the simulations obtained by the vertically distributed model, represening major hydraulic and contaminant transport processes, are in overall agreement with leachate data documented in various studies. However, additional experimental work is required in order to identify and refine the description of basic processes occurring in the landfill.

3) Organic leachate composition. Using the general framework of the models of inorganic leachate composition, the organic leachate composition models add the action of microbial digestion of organics present in landfill moisture, and the movement of oxygen in the landfill. The single well-mixed reactor model, presented in Figure 39, depicts these and other relationships. Separate aerobic and anaerobic microbial populations are assumed to utilize solubilized organic material, creating increased cell mass and gases of decomposition. Active cells are subject to a death rate. Dissolved oxygen in the landfill moisture is utilized by aerobic digestion, and may be replenished by the dissolved oxygen present in influent moisture and by transfer of oxygen from the gas within the landfill itself. The model assumes no direct microbial utilization of solid organics occurs; organics must become dissolved and mixed in the moisture before digestion can take place. Thus the model represents, "the processes of mixing and convection of several contaminants in the landfill moisture, interactions between them imposed by microbial action, the generation of organics into landfill moisture, depletion of solid organics, and oxygen exchange" (Straub, 1980). Each of these relationships is described by mathematical expression. Figure 40 shows the measured and simulated leachate COD for the single reactor model.

A model of four cascaded well-mixed reactors as in Figure 41 was developed as an extension of the single reactor model. Each reactor represents one quarter of the depth of the landfill. The material convected out of reactor i equals the material

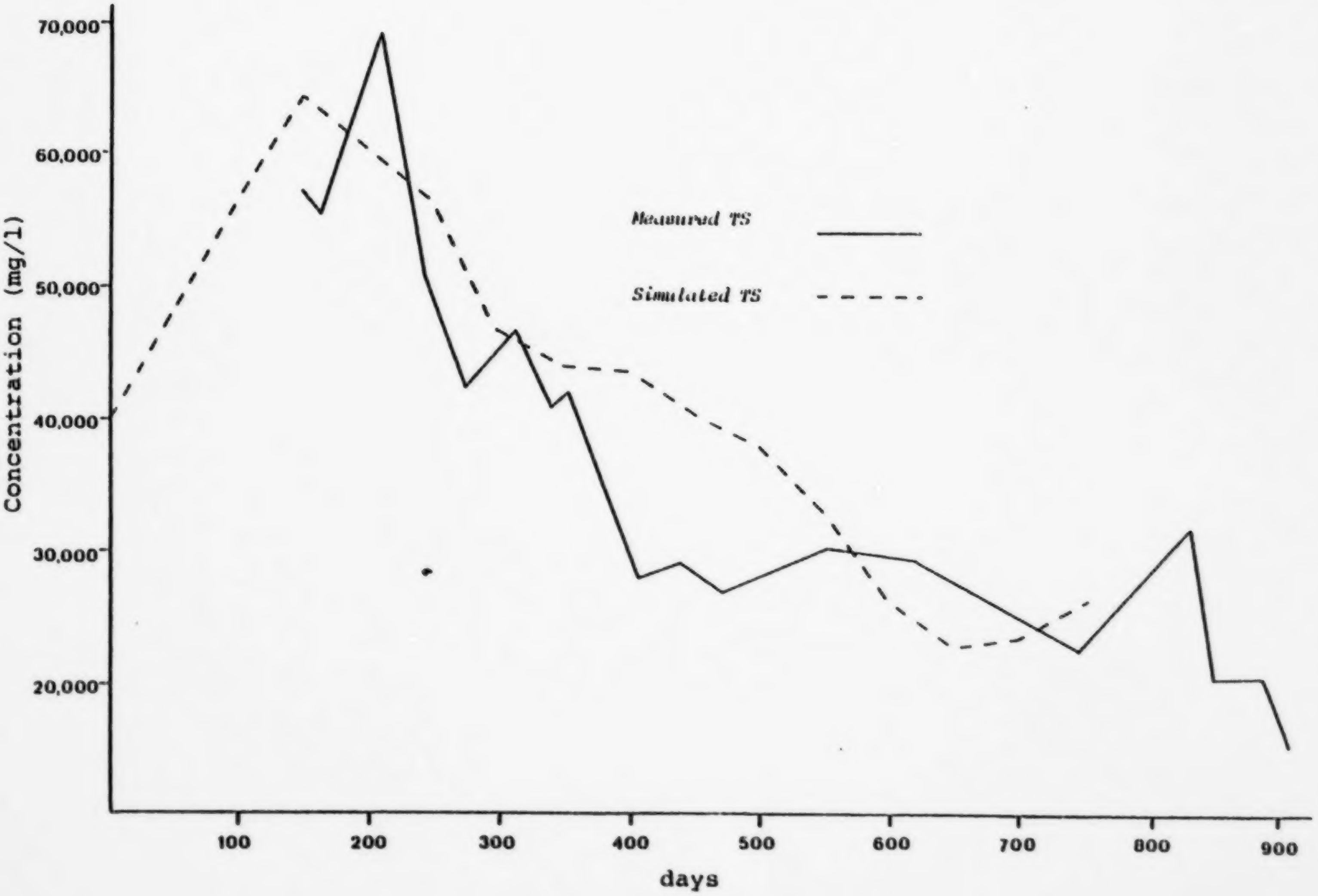


Figure 38. Measured and simulated results of total solids concentration history from Straub and Lynch (1979)

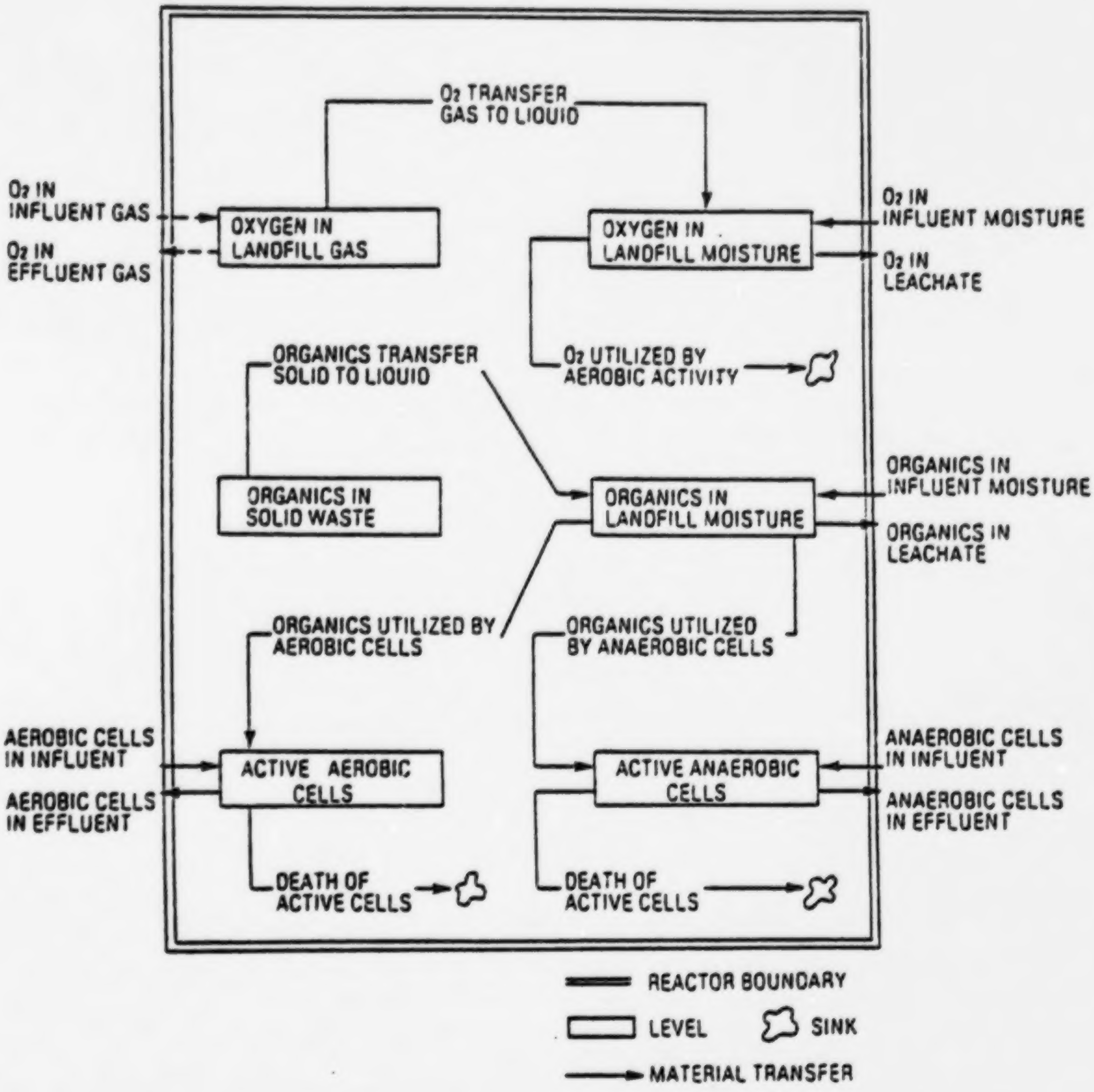

O2 TRANSFER GAS TO LIQUID
O2 IN INFLUENT GAS
O2 IN EFFLUENT GAS
OXYGEN IN LANDFILL GAS
OXYGEN IN LANDFILL MOISTURE
O2 IN INFLUENT MOISTURE
O2 IN LEACHATE
O2 UTILIZED BY AEROBIC ACTIVITY
ORGANICS TRANSFER SOLID TO LIQUID
ORGANICS IN SOLID WASTE
ORGANICS IN LANDFILL MOISTURE
ORGANICS IN INFLUENT MOISTURE
ORGANICS IN LEACHATE
ORGANICS UTILIZED BY AEROBIC CELLS
ORGANICS UTILIZED BY ANAEROBIC CELLS
AEROBIC CELLS IN INFLUENT
AEROBIC CELLS IN EFFLUENT
ACTIVE AEROBIC CELLS
ACTIVE ANAEROBIC CELLS
ANAEROBIC CELLS IN INFLUENT
ANAEROBIC CELLS IN EFFLUENT
DEATH OF ACTIVE CELLS
DEATH OF ACTIVE CELLS
REACTOR BOUNDARY
LEVEL
SINK
MATERIAL TRANSFER

Figure 39. Single well-mixed reactor for model of leachate organics (Straub and Lynch, 1982b)

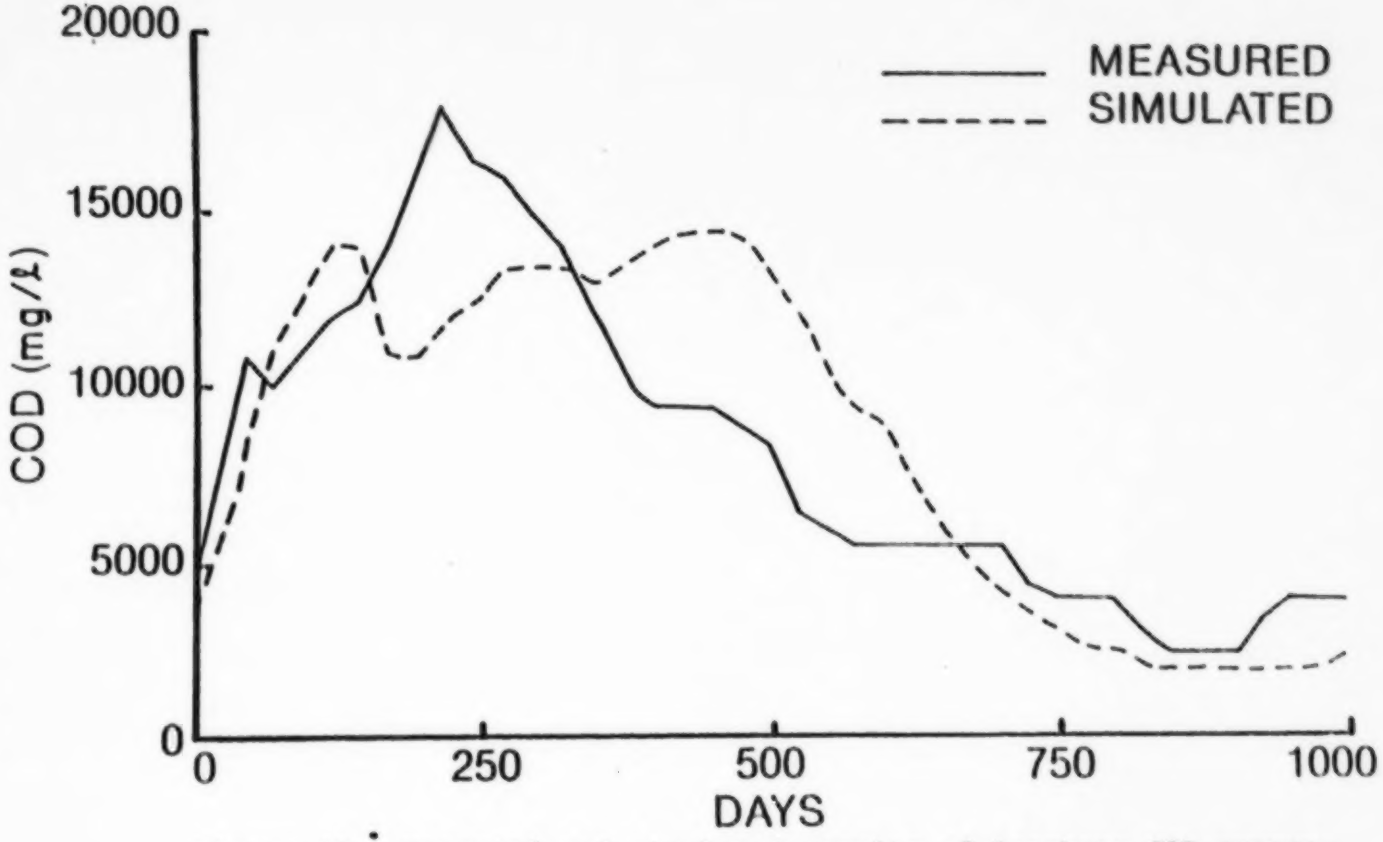
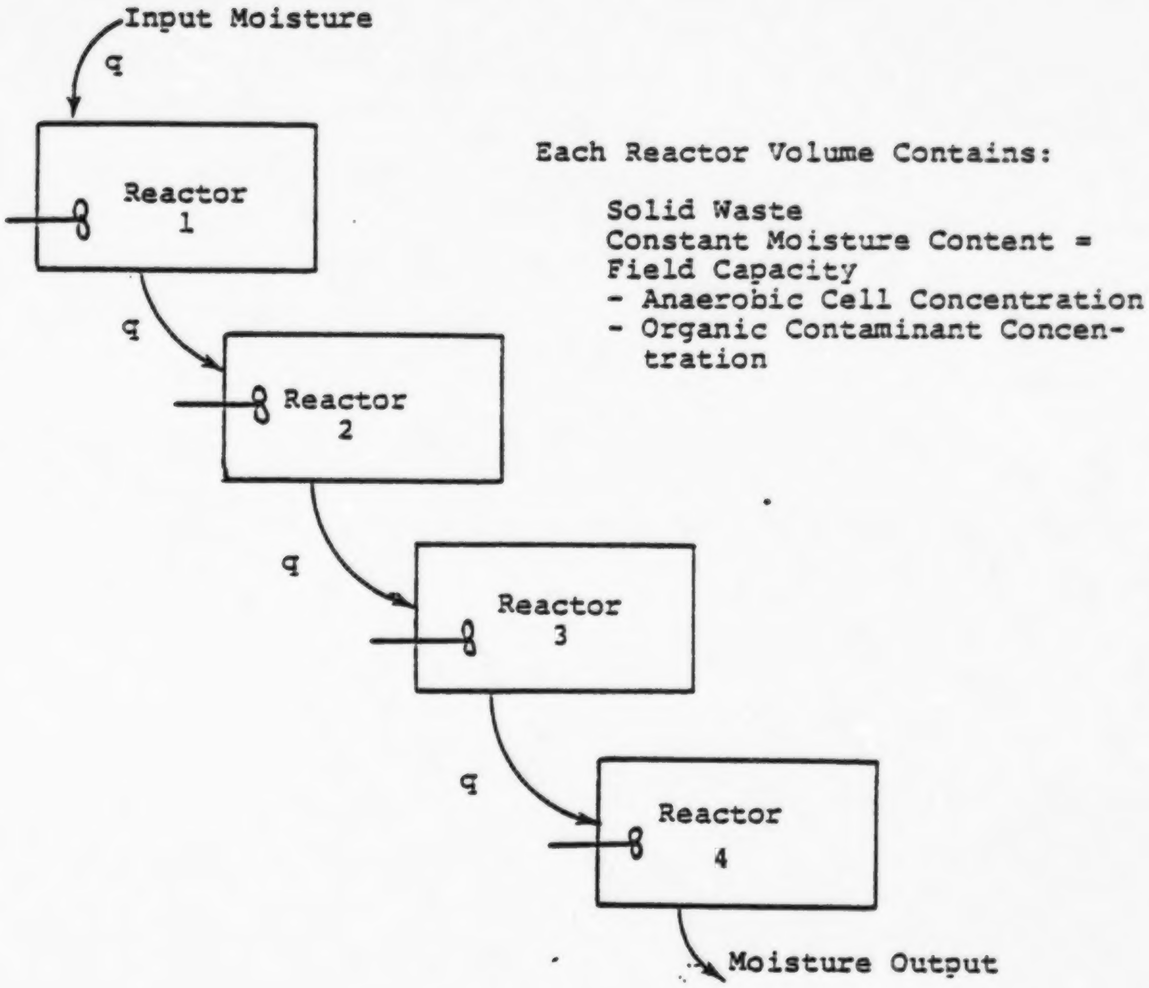


Figure 40. Measured and simulated results of leachate COD concentration history for single reactor model from Staub and Lynch, 1982b)


Input Moisture
q
Reactor 1
q
Reactor 2
q
Reactor 3
q
Reactor 4
Moisture Output

Each Reactor Volume Contains:

Solid Waste
Constant Moisture Content = Field Capacity
- Anaerobic Cell Concentration
- Organic Contaminant Concentration

Figure 41. Vertically cascaded reactor model of landfill (Straub, 1980).

convected into reactor i + 1. Processes within each reactor include: the generation of organics from solid waste to moisture, utilization of organics by anaerobic cells, and convection of organics and cells. This model is thought to have greater general capability, but requires greater empirical support in the development of hydraulic and contaminant dispersion processes. The model is mathematically portrayed as a series of partial differential equations, which are derived by Straub (1980). Figure 42 compares the simulated leachate COD with measured COD from Pohland (1975).

The success with which the models described above simulate actual leachate concentration histories demonstrates the feasibility of analyzing the solid waste leaching as physical/chemical and microbial process models. Reactor models with simplifying assumptions of moisture content, moisture flow, and mixing have been shown to be useful in explaining observed data. These models provide a framework by which the dynamics of leachate volume and quality are described. Because of their complexity, consideration of details of specific situations may be given with models based on the partial differential equations of unsaturated moisture flow and contaminant transport in porous media. Layered effects, vertical distribution of materials and media properties, hydraulic phenomenon, and moisture and contaminant source/sink effects can be reflected in these models. The systematic prediction of leachate behavior and landfill stabilization for particular landfill configurations and designs is provided by detailed simulation of continuous flow of moisture and contaminants. However, as model complexity increase and greater detail is simulated, empirical support required for the formulation of the model also increases. Straub and Lynch (1982a) claim that although the models demonstrate promising predictive abilities, their application to many situations will require a broadened empirical base.

Summary

Table 43 presents a general summary of the leachate composition models previously discussed. At present, leachate composition models are appropriate primarily for research purposes, insofar as mathematical expressions developed to describe the leaching process or concentration histories merely interpret experimental results. Additional experimental work is required in order to identify and refine the description of basic processes occurring in the landfill environment. Eventually, the process models could be applied to field scale problems, although such applications would require extensive empirical support.

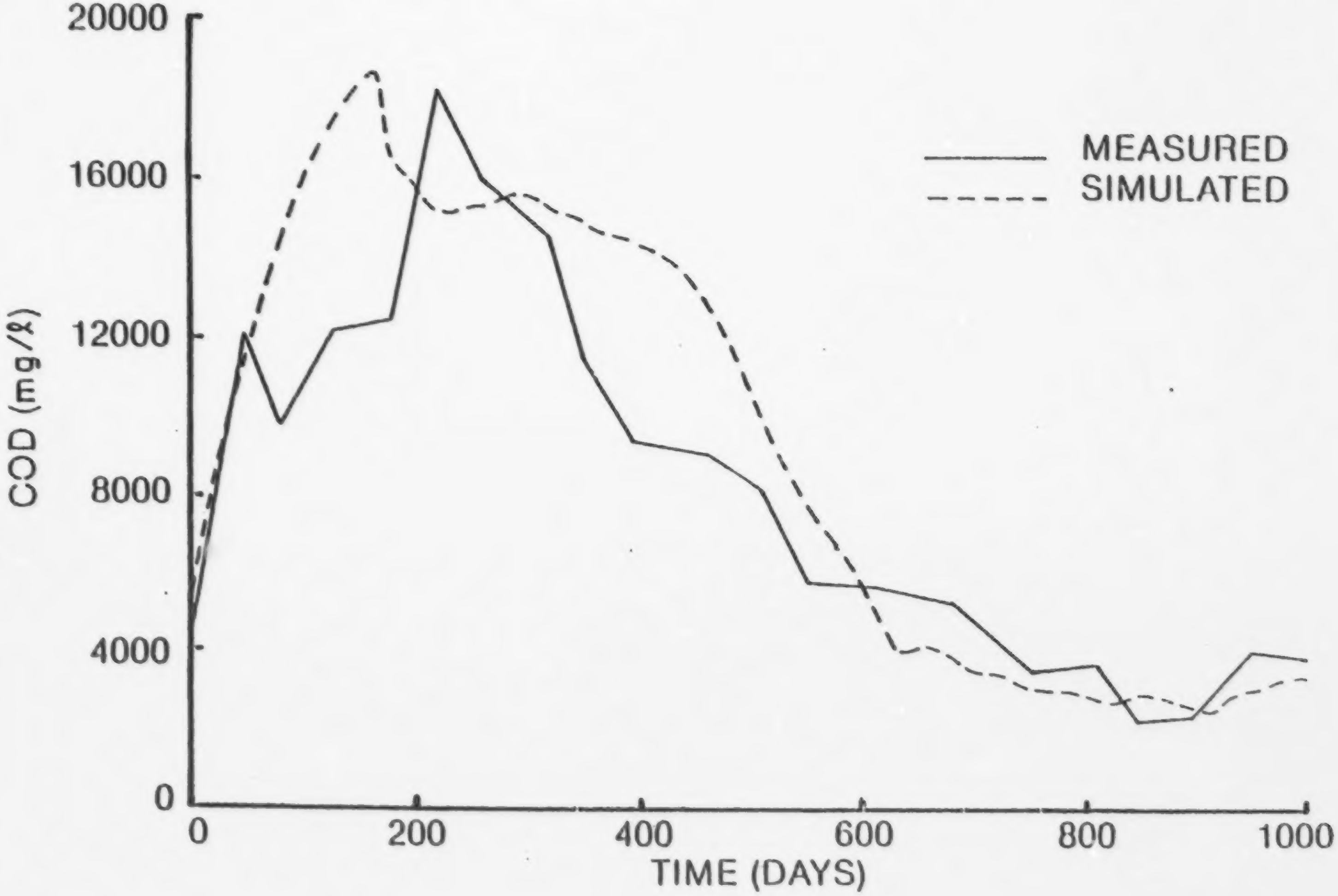


Figure 42. Measured and simulated results of leachate COD concentration history for vertically cascaded reactor model from Staub and Lynch (1982b)

TABLE 43. SUMMARY OF LEACHATE COMPOSITION MODELS

Researcher(s)	Model Concepts	Assumptions	Model Responsiveness To Landfill Characteristics	Agreement of Model With Experimental Data
Qasim and Burchinal (1970)	The theory of column operation and mass transfer is applied to the leaching of chloride from experimental landfill columns. The leaching of chloride over time, among other leachate constituents, was best simulated by desorption breakthrough curves. The concentration histories of 14 other contaminants were related to chloride estimates, with reasonable agreement.	1. The void space of packed refuse is equal to some percentage of the height of the fill. 2. The leaching liquid is in equilibrium with the solid being leached at points of the bed.	Model responsive to the depth of the refuse, refuse compaction, and the cumulative volume of leachate generated by the landfill.	Fair agreement. Deviations from experimental results generally ranges between 5% and 30% over a 163 day testing period at one selected depth.
Revah and Avnimelech (1979)	The decline of concentrations of organic carbon, volatile acids, TKN, NH$_4$, NO$_3$, Fe, and Mn over time were described through an exponential function: $C = ab^t$ where : C = the concentration of the constituent t = time a = concentration at t = o b = the exponential base	1. Peak concentrations of leachate constituents occur at the onset of leaching. 2. Leachate constituent concentrations decline according to exponential decay curves.	Model responsive to refuse density and the effects of shredding as they effect contaminant concentrations as a function of time since the onset leaching.	Fair agreement. General similarity between field data and those obtained by this model.
Wigh (1979)	The concentration history of various leachate constituents was described through a semi-empirical equation. The equation is based on two consecutive first-order reactions having the form $C = \frac{k_1k_2M}{k_2-k_1}\ (e^{-k_1V} - e^{-k_2V})$	1. The total leachable mass of refuse is represented by $M = C_{1/k_2}$ where : c_1 = an experimentally calculated constituent concentration k_2 = an experimentally derived rate constant	Model responsive to cumulative volume of leachate generated and total leachable mass of refuse, but only for landfills of similar refuse, density, depth, etc.	Good agreement. The equation fits well the pattern of contaminant concentration decreases for cumulative leachate volume collected. However, since the fit is empirical in nature, the equation does not represent a fundamental description of the leaching process.

(Continued)

TABLE 43. (Continued)

Researcher(s)	Model Concepts	Assumptions	Model Responsiveness To Landfill Characteristics	Agreement of Model With Experimental Data
Wigh (Continued)	Where C = pollutant concentration M = total leachable mass per unit surface area V = volume of leachate collected per unit surface area k_1k_2 = empirically fit rate constants	2. The concentration histories of leachate contaminants can be represented by the curves described by consecutive first order reactions		
Phelps (n.d.)	This model utilizes mass transfer equations based on flow through a stagnant film on the refuse particles. The model is built upon both theoretically and empirically derived parameters. To test the model, calculated concentration values are fit to observed values for six test results: BOD, total residue, Ca, Fe, COD, and Cl. The experiments used to test the model cover a six-fold range in both refuse depth and infiltration rate, two sizes of refuse particles, and the presence or absence of septic tank sludge.	1. The initial moisture content of the refuse is low enough that the first leachate to appear can be assumed to have travelled the full depth of the refuse. 2. The fluid velocity in the refuse is zero before the wetting front reaches a given depth, and constant and uniform for all points at that depth after the wetting front has arrived. 3. The rate at which soluble material (mass) is transferred from the solid phase to the liquid phase is proportional to the difference in concentrations between these phases. 4. The depth of the refuse does not change with time. 5. The leachate can be considered a dilute solution--changes in leachate concentration are assumed to not effect its density and hence its velocity.	Pollutant concentrations in leachates are described as a function of time, leachate flow rate, and the depth of the refuse.	Fair agreement. Correlation coefficients between the calculated and observed curves for concentration vs. time exceeded 0.9 (R^2). The calculated curves generally agree with the data from the time leachate first appeared to about twice that time. For longer times, the observed concentration values are consistently above those calculated.

(Continued)

TABLE 43. (Continued)

Researcher(s)	Model Concepts	Assumptions	Model Responsiveness To Landfill Characteristics	Agreement of Model With Experimental Data
Phelps (Continued		6. A constant moisture infiltration rate is assumed.		
Straub and Lynch (1982a)	Two models are proposed which portray the concentration histories of leachate inorganics as a function of time since leachate appearance. One model presents refuse as a single well-mixed reactor, providing mass balances for both moisture and contaminant yields. The other is a vertically distributed model of contaminant transport in an unsaturated porous medium. Calculated concentrations are compared with empirical data from several pertinent studies.	For the single well-mixed reactor model: 1. Landfills can be portrayed as a single well-mixed reactor. 2. Contaminant concentrations of infiltrate are zero. 3. Refuse moisture content remains essentially constant at field capacity. 4. Internal leachate generation is negligible. For the vertically distributed unsaturated flow model: 1. Landfill moisture flux is by gravity drainage only, i.e., no moisture diffusion and no contaminant dispersion/diffusion out of the system occurs. 2. Concepts of hydraulic and contaminant transport in soils can be applied to solid waste.	For the single well-mixed reactor model, model parameters include moisture infiltration rate, volume, depth, and field capacity of refuse. For the vertically distributed-unsaturated flow model, model parameters include leachable contaminant mass, refuse moisture content, volume, depth, density and field capacity of refuse and moisture application rates.	Very good agreement. Simulated vs experimental results show the leaching of inorganic contaminants is explainable in terms of fundamental transport processes.

(Continued)

TABLE 43 (Concluded)

Researcher(s)	Model Concepts	Assumptions	Model Responsiveness To Landfill Characteristics	Agreement of Model With Experimental Data
Straub and Lynch (1982b)	The concentration histories of leachate organics are modeled, three models were developed, based on an unsaturated flow of moisture contaminant generation and transport, oxygen exchange, and microbial activity. One is a single well-mixed reactor model similar to that described above. The second is a vertically cascaded well-mixed reactor model. The third incorporates unsaturated flow and contaminant transport modeling, described above, to simulate organic leachate strength. Calculated concentrations are compared with empirical data from several pertinent studies.	Each of the reactor models makes simplifying assumptions regarding moisture storage, moisture flow, and contaminant distribution in order to focus on contaminant generation, microbial activity, oxygen exchange, and dilution/convection effects.	Each of the reactor models is responsive to landfill volume, void volume, moisture content at field capacity, moisture flow rates, concentrations of substances in moisture entering the landfill, ultimate mass of leachable constituents per unit bulk volume of refuse, and an assumed maximum constituent concentration.	Very good agreement. Each of the models closely approximates experimental COD concentration histories, but the unsaturated flow and contaminant transport model is more precise, and is thought to have greater field scale potential.

SECTION 5

LEACHATE MIGRATION

INTRODUCTION

In this section, interactions between leachate and underlying soils are addressed. An extensive review and analysis of the movement of leachate contaminants through underlying soils has been performed by Weston, Inc. (1978). This section summarizes and provides supplementary information to the findings of this study.

Soil is a dynamic physical, chemical, and biological system which may be envisioned as material containing biodegradative matter and chromatographic properties, with numerous finite variables that influence leachate migration. For instance, inorganic species may be adsorbed or complexed by soil components or leached from the surface of soil particles. Organic substances may be ultimately mineralized, yielding inorganic substances, harmless gases, and water. Organic matter accumulates as resynthesized substances of microbial origin and as degraded organic residue (Alexander, 1971; Fuller, 1978; McCarty and Rittman, 1981).

The attenuation and migration of contaminants in the soil/water system are influenced by the following physical, chemical, and biological mechanisms:

- Diffusion and dispersion;
- Dilution;
- Straining;
- Precipitation/dissolution;
- Adsorption/desorption;
- Complexation;
- Ion exchange;
- Redox; and

- Microbial activity.

Although these mechanisms can be grouped as above, clear distinctions cannot always be made because of the simultaneous occurrence of numerous interactive reactions. Moreover, a factor which may mobilize or attenuate a constituent in one soil, may have no effect in another soil; a factor which may inhibit movement of one element may have no effect on the movement of another element. Each of the preceeding phenomena will be discussed in the following subsections in relation to the effects on the applicable physical, chemical, and biological mechanisms.

The multitude of reactions that are operative when contaminants enter a soil can be, as listed above, identified with reasonable assurance, but quantitative data relating to the attenuation of individual constituents by specific mechanisms are not available, consequently, considerable effort has been placed on the development of sophisticated leachate migration models. However, due to the highly complex leachate/soil environments, no leachate migration model exists that can simulate all of the physical, chemical, and biological processes occurring in a typical landfill system. In this section, emphasis is placed on a discussion of existing conceptual-mathematical models, since these models appear the most promising, but, unfortunately, also the most complex ones for predicting the performances of waste disposal sites. An extensive list of available simulation models is given, and the advantages and disadvantages of these models for simulating landfill behavior are discussed. Design examples are not presented because of the large number and complexity of simulation models. Interested designers and investigators are referred to the original literature for detailed discussion and methodology.

LEACHATE/ENVIRONMENTAL INTERACTIONS

Soil Properties

The soil is a heterogenous, polydisperse system of solid, liquid, and gaseous components in various proportions. The solid component of the soil is made up of primary minerals, clay minerals, and hydrous oxides, together with organic matter and living organisms, forming a polyphase system of more or less discrete particles or aggregates. Separate particles vary widely in size and shape. The finest particles are clay--2μ or less in diameter, which constitute the major portion of the soil colloidal material. The organic matter, particularly humus, and clay, affect many of the properties of soil, especially surface phenomena (Bear, 1955; Stumm and Morgan, 1981). In this heterogenous system, the soil solution acts as the medium by which chemical reactions between members of the different phases and of the same phase are made possible. In the USDA system, the soil texture designation (e.g., sandy loam, silty clay), is based on the amounts of sand-, silt-, and clay-sized particles in the soil. The triangular diagram used to make this textural classification of the soils in the USDA system is shown in Figure 43.

Soils are very porous bodies. About 40 percent of the volume of dry, compact sand and about 60 percent of dry, compact highly organic soils is air. By volume, about 50 percent of the most common soils (loams, silt loams, and fine sandy loams) consist of pore spaces which are of a cellular nature (Walkman, 1952; Alexander, 1971). This porosity is an important quality that affects the movement of leachates in soils and determines biological activities, the exchange of gases, chemical reactions, and biological changes.

Soils are chemically sorbent bodies consisting of: (1) inert chemical compounds, (2) difficult and easily soluble substances, (3) soluble salts and acids, and (4) complex insoluble compounds assessing the property of exchanging their basic elements for other basic elements and hydrogen. Soil also may contain a wide variety of organisms as shown in Table 44 (Mitchell, 1974). Investigations have discovered 320,000 to 500,000 CFU[*] bacteria per gram of sandy soil material, and 360,000 to 600,000 CFU bacteria per gram of loam. In soils that are rich in organic matter, as many as 2,000,000 to 200,000,000 CFU bacteria per gram of material have been found (Walkman, 1952; Alexander, 1971). In a complex physical and chemical soil medium, organisms pursue their activities not as individual groups, but as a population. A soil does not represent a homogeneous biological entity, but rather a media where a series of complex biological activities are occurring simultaneously.

The soil properties most useful in predicting mobility of leachate contaminants are: (1) texture (clay content); (2) content of hydrous oxides (Fe, Mn, and Al); (3) type and content of organic matter; (4) particle size distribution; (5) cation exchange capacity; and (6) soil pH. Relative mobilities of leachate contaminants through soils are quite variable. However, a qualitative prediction is possible if the important soil physical and chemical properties are known (Korte, et al., 1975; Fuller, 1976; Fuller, 1978; Griffen and Shimp, 1978; Weston, 1978). In the following discussion, specific examples are used to illustrate the various reactions which influence leachate mobility and attenuation.

Migration Mechanisms

Physical Mechanisms--

The most important physical properties of a soil system relative to leachate migration are diffusion and dispersion,

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[*] Colony forming unit

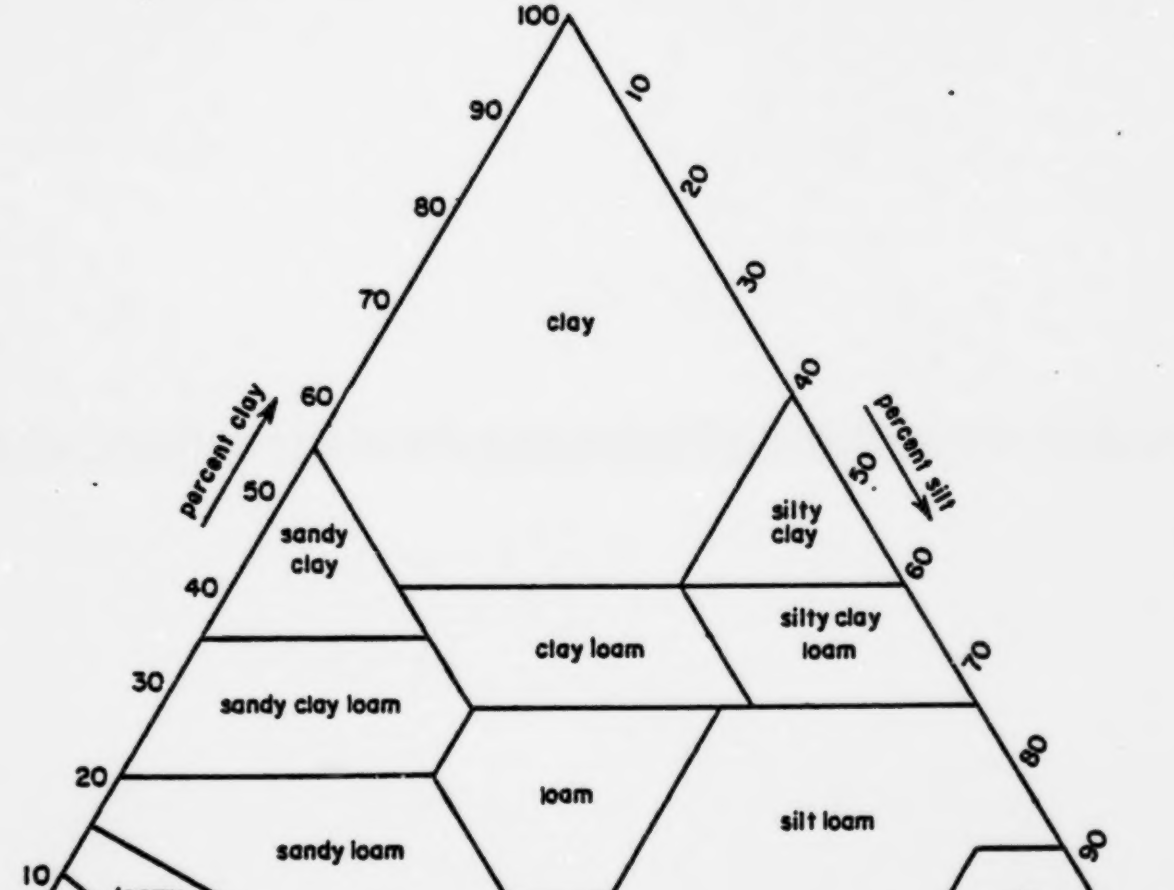
100
90
80
70
60
50
40
30
20
10
percent clay
10
20
30
40
50
60
70
80
90
100
percent silt
100
90
80
70
60
50
40
30
20
10
percent sand
clay
sandy clay
silty clay
clay loam
silty clay loam
sandy clay loam
loam
silt loam
sandy loam
loamy sand
sand
silt

Figure 43. USDA soil textural classification (U.S. Soil Conservation Service, 1971).

TABLE 44. SOIL ORGANISMS (after Mitchell, 1974)

Soil Organisms	Protists	Bacteria	Heterotrophic 1. Nitrogen-fixing a. Symbiotic b. Nonsymbiotic 2. Bacteria that require combined nitrogen.
			Autotrophic Bacteria that oxidize nitrogen, sulphur, iron, hydrogen, and carbon compounds, and sulphur and/or hydrogen.
		Fungi	Yeasts Molds Actinomycetes
		Algae	Blue-green Diatoms Grass-green
		Protozoa	
	Animals	Nematodes Rotifers Worms Insects, etc.	

dilution, physical sorption, and straining.

Diffusion and dispersion effects--Molecular diffusion is caused by the concentration gradient of contaminants, that is, the movement of a chemical component from a region of high concentration to one of low concentration. In the case of very low leachate flow rates, diffusion of soluble species in the soil solution may be a significant migration mechanism (Mang, *et al.*, 1978).

Hydrodynamic dispersion is the result of variations in pore velocities within the soil. Within a given soil pore, the flow rate is slower near the walls than in the center of the pores. The soil water (or leachate) also flows faster in the larger pores than in small pores. Hydrodynamic dispersion is effective in attenuating the maximum constituent concentration rather than total qualtity of the constituent in a pulse or slug of leachate. For large leachate inputs, hydrodynamic dispersion will not be an effective mechanism (Ogata, 1970; Weston, 1978).

Dilution effects--When leachate flows from the unsaturated zone to the zone of saturation below the water table, dilution of the contaminant concentrations occur. The rate of contaminant dilution is proportional to the solution flux of both leachate and ground water. The finer the soil texture the slower the leaching solution tends to flow, and therefore, the slower the rate of dilution. The leachate flow rate is also directly proportional to the moisture content of the soil through which leachate is being transported. If the region of soil between the water table and the bottom of a landfill is unsaturated, the vertical transport rate of the leachate from the disposal site will be orders of magnitude smaller than that when soil is saturated (Ham, 1975).

Physical sorption effects--Physical sorption is a function of Van der Waals forces, hydrodynamic, and electrokinetic properties of small soil particles. As mentioned by Fuller (1977), only a small portion of the reactions of trace contaminants in soil/water solutions can be defined as physical adsorption; however, physical adsorption is an important removal mechanism as regards bacteria and viruses (Gilbert, 1976). The remainder of the sorbed contaminants can be ascribed to chemical bonding or chemical adsorption, as will be described later.

Straining effects--Suspended solid particles in solutions generally range in size from 0.001μ to 8μ. The removal of constituents by soil filtering is applicable to a wide range of particle sizes, and not exclusively, to larger particles. Another straining effect by soils is the attenuation of nonpolar organic compounds (such as oils, grease, and hydrocarbons of lower molecular weight). Even a small depth of soil can remove a large quantity of such material (Mang, *et al.*, 1978).

Chemical Mechanisms--

Five types of reactions occur which are basically chemical in nature: (1) precipitation/dissolution, (2) adsorption/desorption, (3) complexation, (4) ion exchange, and (5) reduction/oxidation (redox).

Precipitation/dissolution reaction--Precipitation/dissolution reactions can control concentration levels and limit the total amount of contaminants in leachates when leaching through soils. The migration trends of contaminants are usually toward an equilibrium state, controlled by the solubilities of solids. For example, when the soluble concentration of a contaminant in leachate is higher than that of the equilibrium state, precipitation occurs, thus the contaminant level of the resulting leachate is attenuated.

Precipitation/dissolution reactions are especially important in trace metal migration in the soils (Garrels and Christ, 1965; Santillan-Medrano and Jurinak, 1975; Griffin, et al., 1976; Leckie and James, 1976; Weber and Posselt, 1976; Griffin and Shimp 1978). The important solubility products of common trace metal compounds are presented in Table 45. In general, cadmium, copper, lead, nickel, and zinc can form carbonate solids in relatively oxidizing leachate environments, and sulfide solids in relatively reducing environments where soluble sulfide is present. Iron and manganese can form hydroxide or oxide solids under oxidizing environments and sulfide solids when soluble sulfide is present. The attenuation effects of precipitation/dissolution of metals is more significant at neutral, or higher than neutral pH values. At lower pH levels, other effects (such as adsorption) will usually prevail. Since precipitation/dissolution and sorption occur simultaneously in the soil, it is difficult to separate the two processes.

Adsorption/desorption reaction--Sorption (adsorption or desorption) is the most common mechanism generally associated with trace contaminant migration in soils. Under oxidizing or low pH conditions, adsorption can usually regulate the concentration of a contaminant well below the level controlled by precipitation. The high anion retention of soils that cannot be attributed to anion exchange or precipitation, can usually be attributed to adsorption (Jenne, 1968). Because of the relatively rapid rate of adsorption in comparison to precipitation reactions, adsorption mechanisms may affect the concentration of a constituent even if that constituent has a low solubility controlling solid.

Co-precipitation is the precipitation of an otherwise soluble substance along with an insoluble precipitate. These two substances may precipitate simultaneously or one may follow the other (Peters, et al., 1974). The co-precipitation of a constituent as an impurity in solid phase accumulations and

TABLE 4 5. IMPORTANT SOLUBILITY PRODUCTS (Ksp)† OF TRACE METALS. (Sillen and Martell, 1964, 1971; Ringbon, 1973; Stumm and Morgan, 1981)

Metal	Oxide	Hydroxide	Carbonate	Sulfide	Chloride	Phosphate	Silicate
Cd(II)		13.6	13.6	26.1			
Ca(II)		5.26	8.32 (calcite), 8.22 (aragonite), 16.7† (dolomite)	2.94		6.25 ($CaHPO_4$) 26 ($Ca_3(PO_4)_2$) 1.14 ($CaH_2(PO_4)_2$) 6.4 ($CaHPO_4(H_2O)_2$) 48.9 ($Ca_4H(PO_4)_3$) 44.6 ($Ca_5OH(PO_4)_3$) 120.8 ($Ca_{10}(PO_4)_6F_2$)	3.7 ($CaSiO_3$), 52.3† (anorthite), 585 (ca-montmorillonite)
Cu(II)	20.4	18.6	9.63 ($CuCO_3$), 33.2 ($Cu_2CO_3(OH)_2$)	35.2		37.7	
Fe(II)		15.3	10.2	16.9(FeS)		33.3	18.9†
Fe(III)	80† (Fe_2O_3)	39.3		18.2 (Fe_3S_4)		25.8	
Pb(II)	15.4 (PbO)	16.1 18.8 ($Pb_3(OH)_2(CO_3)_2$)	13.1	26.6	4.79	43.5, 12.6† ($PbHPO_4$)	
Mg(II)		9.2 (active, 11.6 (brucite)	4.9 (magnesite), 5.4 (nesquehonite), 16.7 ($MgCa(CO_3)_2$)		4.44† ($MgCl_2(H_2O)_6$), 4† ($KMgCl_3(H_2O)_3$)	28.4 ($Mg_3(PO_4)_2$) 12.6† ($MgNH_4(PO_4)$) 13.2† ($MgNH_4PO_4(H_2O)_6$ 5.0† ($MgHPO_4(H_2O)_3$	

(Continued)

TABLE 4.5(Continued)

Metal	Oxide	Hydroxide	Carbonate	Sulfide	Chloride	Phosphate	Silicate
Mn(II)	0.92[†]	12.7	9.3	12.9 (crystaline), 15.7 (precipitated)		22	13.2[†]
Hg(II)	25.7[†]	25.4		52.2 (Meta-cinnabar) 53.6 (cinnabar)	13.8		
K(I)			-4.11				76[†] (orthoclase) 124[†] (muscovite)
Na(I)							40.6[†] (albite) 294[†] (Na-montmorillonite)
Zn(II)		15.7 (Amorphous), 16 (Amorphous, aged), 16.9 (cryst. aged)	10.8	25.2 (sphalerite), 22.0 (wurzite), 22.1 (precipitated)		36.7	21.02[†]

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[*] Values in p^{K}sp (log K_{sp}) when I (ionic temperature) = 0, T (temperature) = 25°C

[†] NaAlSi$_3$O$_8$(s, albite) + 7H$_2$O + H$^+$ = Al^{3+} + Na$^+$ + 3H$_4$SiO$_4$ + 3OH$^-$;
CaAl$_2$Si$_2$O$_8$(s, anorthite) + 8H$_2$O = 2Al^{3+} + 2H$_4$SiO$_4$ + 8OH$^-$ + Ca^{2+};
1 1/2 KAlSi$_3$O$_8$(s, orthoclase) + 12H$_2$O = 1 1/2 Al^{3+} + 1 1/2 K$^+$ + 4 1/2 H$_4$SiO$_4$ + 6OH$^-$;

(Continued)

TABLE 45. (Concluded)

$KAl_3Si_3O_{10}(OH)_2(s, \text{muscovite}) + 10H_2O = 3Al^3 + k^+ 3H_4SiO_4 + 10OH^-;$

$3Na_{0.33}Al_{2.33}Si_{3.67}O_{10}(OH)_2(s, \text{Na-montmorillonite}) + 30H_2O = 7Al^{3+} + 11H_4SiO_4 + 22OH^- + Na^{-1};$

$3Ca_{0.33}Al_{4.67}Si_{2.33}O_{20}(OH)_4(s, \text{Ca-montmorillonite}) + 60H_2O = 14Al^{3+} + 22H_4SiO_4 + 44OH^- + Ca^{2+};$

$CaMg(CO_3)_2\ (s, \text{dolomite}) = Ca^{2+} + Mg^{2+} + 2CO_3^{2-};$

$Fe_2O_3(s) + 3H_2O = 2Fe^{3+} + 6OH^-;$

$FeSiO_3(s) + H_2O = Fe^{2+} + 2OH^- + SiO_2(s);$

$Pb_3(OH)_2(CO_3)_2(s) = 3Pb^{2+} + 2OH^- + 2CO_3^2;$

$PbO(s) + H_2O = Pb^{2+} + 2OH^-;$

$PbHPO_4(s) = Pb^{2+} + HPO_4^{2-};$

$MgNH_4\ (PO_4)(s) = Mg^{2+} + NH_4^+ + PO_4^{3-};$

$MgNH_4(PO_4)(H_2O)_6(s) = Mg^{2+} + NH_4^+ + PO_4^{3-} + 6H_2);$

$MgHPO_4(H_2O)_3(s) = M_g^{2+} + HPO_4^{2-} + 3H_2O;$

$MgCl_2(H_2O)_6(s, \text{bischofite}) = Mg^{2+} + 2Cl^- + 6H_2O;$

$KMgCl_3(H_2O)_3(s, \text{carnallite}) = K^+ + Mg^{2+} + 3Cl^- + 3H_2O;$

$MnO_2(s) + 2H^+ = Mn^{2+} + 1/2\ O_2 + H_2O;$

$MnSiO_3(s) + H_2O = Mn^{2+} + 2OH^- + SiO_2(s);$

$HgO(s) + H_2O = Hg^{2+} + 2OH^-;$

$ZnSiO_3(s) + H_2O = Zn^{2+} + 2OH^- + SiO_2(s)$

crystal structures is common in nature. Since it is extremely difficult to separate the effects of co-precipitation and adsorption, both are usually included as adsorption. Some of the most important adsorption mechanisms in soils are those associated with iron and manganese hydrous oxides (Jenne, 1968; Gadde and Laitinen, 1974; Korte, *et al.*, 1975; Shuman, 1976; Kinniburgh, *et al.*, 1976; Fuller, 1976; Shuman, 1977; Fuller, 1977; Fuller, 1978). Jenne (1968) has suggested that hydrous oxides are the main controlling factor in the environmental fixation of trace metals. This theory is supported by findings that manganese and iron oxide contents are highly correlated with the quantity of trace metals in soils (Taylor and McKenzie, 1966; Korte, 1976).

Clay minerals, hydrated aluminum oxides and soil-organic solids may also adsorb constituents from the solution phase of soil/water system (Korte, *et al.*, 1975; Fuller, 1976; Griffin, *et al.*, 1976; Kinniburgh, *et al.*, 1976; Fuller, 1977; Roulier, 1977; Fuller, 1978; Griffin and Shimp, 1978; McBride, 1978). As in the case of precipitation reactions, sorption processes are strongly dependent on pH. Because of pH-dependent charge characteristics, soils can exhibit either cation or anion exchange characteristics, (Jenne, 1968; Frost and Griffin, 1977; Shuman, 1977; Griffin and Shimp, 1978; Stumm and Morgan, 1981).

Complexation--Complexation involves the reaction of metal ions with inorganic anions and organic ligands (chelation) to form complex inorganic ions and organometallic complexes (Stumm and Morgan, 1981). Complex formation may affect attenuation and migration of constituents in two ways. In the solution phase, it can greatly increase the concentration of constituents by the formation of soluble complex ions. On the other hand, if the complex formation exists between the soluble constituents and solid surfaces, especially organic chelates, then the levels of constituents in the resulting leachates can be decreased (Schnitzer, 1971; Stevenson, 1976; Mattigod and Sposito, 1977; Doner, 1978; Mang, *et al.*, 1978). The concentrations of complex species in leachate are not necessarily the result of equilibrium processes. Species which are thermodynamically unstable but which, due to kinetic constraints, exist in significant amounts are called nonlabile or inert (Siegel, 1971). Because of these nonequilibrium processes, extrapolation of laboratory determined thermodynamic stabilities to natural conditions requires considerable caution (Ruben, 1974). Although the "apparent" stability of metal ions and natural organics have been investigated in a few cases, the extent of the interaction of these organic compounds with metal ions in the real environment is not known (Malcolm, 1970; Schnitzer, 1971). Since both soil and municipal refuse contain a wide variety of organic compounds and complexes, the net attenuation or migration of complex species, is very difficult, if not impossible, to evaluate with any degree of certainty.

Ion exchange--Most of the ion exchange effects in soil/water system originate from the exchange sites on layered silicate clays and organic matter. In general, the layered aluminum-silicate secondary minerals in soils hold a permanent negative charge. Murrmann and Koutz (1972) pointed out that the cation exchange property arises from the need to balance the negative charge of clay micelles to maintain neutrality. To accomplish this, positive ions in the soil solution become associated with the negative charge on the exchange complex. These ions are mobile and readily exchange with other cations in the soil solution to maintain chemical equilibrium. This process represents cation exchange. The total capacity of soils to exchange cations is called the cation exchange capacity (CEC); the CEC of any particular soils is affected by the kind and quantity of clay mineral and organic matter and the pH of the soil/water solution. For the three predominant types of clay minerals, CEC values can be ranked in the following order (Wilson, 1980): montmorillonite > illite > kaolinite.

Soil textures greatly affect CEC values. Generally, soils containing smaller particles have higher CEC values because of the large specific surface area and correspondingly available exchange sites. The organic matter fraction of a soil may contribute substantially to the CEC by contributing a negative charge to the soil exchange complex. Additions of organic matter through sludge application may improve the CEC of soils with naturally low CEC (Broadbent, 1973). The cation exchange capacity increases with increasing soil pH. This effect is due to an increased negative charge on the soil exchange complex with rising pH as a result of increasing ionization of soil acid groups and decreasing proton addition to the basic groups. The change goes in the opposite direction if the pH decreases. In most cases, the negative charge greatly exceeds the positive charge, resulting in a net negative charge (Stumm and Morgan,1981).

Major cations in landfill leachates (i.e., calcium, magnesium, potassium, and sodium) are usually present in higher concentrations than any of the trace metals. Consequently, trace metals cannot successfully compete for the cation exchange sites that are dominated by the major cations (Wiklander, 1974; Griffin, et al., 1976; Fuller, 1977; Garcia-Miragaya, and Page, 1977). As a result, the removal of trace metals from soil-water interfaces by cation exchange is usually insignificant when compared to other mechanisms (Murrmann and Koutz, 1972).

In nature, sorption, solid-liquid complexation, and ion exchange processes are difficult to distinguish from one another. Although ion exchange mechanisms can be distinguised by the initial and subsequent cation/anion concentrations, these three mechanisms are usually grouped together as if they were one mechanism.

Redox reaction--Redox (reduction/oxidation) reactions of contaminants will occur when conditions of redox potential in leachates are different than that of the soil solutions. As discussed previously, redox reactions are important in the migration of contaminants since they can affect the soluble complexes and solid transformations, subsequently affecting the solubilities of contaminants. In general, for iron and manganese, reduced forms are more soluble than oxidized forms. For other trace metals, however, a reducing environment is more favorable for the attenuation of metals if sufficient sulfide is present for the precipitation of metallic sulfides which are generally very insoluble.

Microbiological Mechanisms--

Microbiological reactions affecting contaminant migration are numerous. The following major effects can be attributed to microbial activities (i.e., biodegradation): redox effects, mineralization, immobilization, precipitation/dissolution, and complexation.

Redox reactions are greatly affected by the biodegradation of organic compounds in soil. The mode of degradation not only changes with the organism specie, but it also may affect the redox potential and thus the oxidation state and chemical forms of all constituents in both solution and solid phases of the soil system.

Through mineralization, plant nutrients, organic chemicals, microbial tissues, and organic-inorganic complexes may be converted into inorganic states. Through biological assimilation, the inorganic nutrients, and trace metals may be transformed into microbial tissue, thus biologically immobilizing these constituents.

Organic complexes which accumulate in soils as a result of both microbial synthesis and degradation have a high capacity to combine strongly with trace metals and other constituents. Through these reactions, the constituents in soil systems can be mobilized, complexed, precipitated, or sorbed.

Migration Trends of Contaminants

General Parameters--

The migration trends of general parameters in the soil/water environments to be discussed are pH, redox potential (Eh), TOC, and alkalinity.

pH--pH levels in leachates are usually acidic (refer to Section 4, Leachate Composition). The result is due to the biodegradation reactions as illustrated by the following two generized equations:

$$(CH_2O)_{106}(NH_3)_{16}H_3PO_4(s) + 53\ SO_4^{2-} \xrightarrow[\text{bacteria}]{\text{Anaerobic}}$$

$$106\ HCO_3^- + 53\ HS^- + 16\ NH_4^+ + HPO_4^{2-} + 39\ H^+, \text{ or}$$

$$(CH_2O)_{106}(HN_3)_{16}H_3PO_4(s) + 106\ O_2 \xrightarrow[\text{bacteria}]{\text{Aerobic}}$$

$$106\ HCO_3^- + 16\ NH_4^+ + HPO_4^{2-} + 92\ H^+.$$

Low leachate pH levels can usually be increased when leachate percolates through underlying soils. This is acomplished by the dissolution of solids in the soils which contain a buffering capacity, such as carbonate, oxide, or sulfide solids. Following are two typical examples:

$$CaCO_3\ (s) + H^+ = Ca^{2+} + HCO_3^-$$

$$M_{\frac{2n}{m}}S_n(s) + nH^+ = \frac{2n}{m}M^{m+} + nHS^-$$

(Where $M_{\frac{2n}{m}}S_n$ represents metallic sulfides)

In general the calcium carbonate ($CaCO_3$) content in soils is the most important factor responsible for the leachate pH increase.

Redox potential--The redox condition of leachates from active biodegradation landfills is usually reducing. Soluble sulfides produced in the landfill are the main species which control the redox level. Numerous redox reactions can be expected during leachate migration through underlying soils. Table 46 lists some important related theoretical redox reactions in the soil/water system. For example, if the pH and redox potential in the soil/water environment is 7 and -200 mV, respectively, then from Table 46, H_2S can begin to undergo transformation to SO_4^{2-} because the Eh of the H_2S-SO_4 redox couple is - 210 mV. Since this Eh value is more negative than the soil/water Eh (-200 mV), there will be a strong tendency for H_2S to be oxidized to SO_4^{2-}. If the original iron solid in the soil is $FeCO_3(s)$, then transformation to $Fe(OH)_3(s)$ also can be seen. (Eh $FeCO_3$-$Fe(OH)_3$ couple = - 385 mV), which tends to reduce the soluble Fe levels. Under these soil/water conditions (pH = 7, Eh = -200 mV), $MnO_2(s)$ could be gradually converted to $MnS(s)$ which would subsequently increase the soluble Mn levels. (Eh $MnO_2(s)$ - $MnOOH(s)$ couple = + 670 mV; Eh $MnOOH(s)$ - $MnS(s)$ couple = + 407 mV). Other redox reactions and their effects can be qualitatively predicted from Table 46. Redox couples not included in Table 46, can be calculated from the thermodynamic data such as listed by Latimer (1952), Sillen and Martell (1964), and Sillen and Martell (1971).

TABLE 4.8. THEORETICAL REDOX CONDITIONS FOR REDOX COUPLES* (Sillen and Martell, 1964, 1971; Ringborn, 1973)

Redox Couple	Reaction	Eh (mV) pH=6	Eh (mV) pH=7	Eh (mV) pH=8
Fe^{3+} — Fe^{2+}	$Fe^{3+} + e = Fe^{2+}$	+764	+764	+764
NO_3^- — $N_2(g)$	$1/5\ NO_3^- + 6/5\ H^+ + e = 1/10\ N_2\ (g)^* + 3/5\ H_2O$	+831	+759	+687
$MnO_2(s)$ — $MnOOH\ (s)$	$MnO_2\ (s) + H^+ + e = MnOOH\ (s)$	+730	+670	+610
$Mn_3O_4(s)$ — $MnCO_3(s)$	$1/2\ Mn_3O_4(s) + 3/2\ HCO_3^-(10^{-2}) + 5/2\ H^+ + e = 3/2\ MnCO_3(s) + 2H_2O$	+760	+610	+460
$MnO_2(s)$ — $MnCO_3(s)$	$1/2\ MnO_2(s) + 1/2\ HCO_3^-\ (10^{-2}) + 3/2\ H^+ + e = 1/2\ MnCO_3(s) + 3/8\ H_2O$	+660	+570	+480
$MnOOH(s)$ — $MnCO_3(s)$	$MnOOH(s) + HCO_3^-\ (10^{-2}) + 2H^+ + e = MnCO_3(s) + 2H_2O$	+670	+550	+430
$MnOOH(s)$ — $Mn_3O_4(s)$	$3MnOOH(s) + H^+ + e = Mn_3O_4(s) + 2H_2O$	+500	+440	+380
NO_3^- — NO_2^-	$1/2\ NO_3^- + H^+ + e = 1/2\ NO_2^- + 1/2\ H_2O$	+489	+429	+369
$MnOOH(s)$ — $MnS(s)$	$MnOOH(s) + HS^-\ (3x10^{-6}) + 2H^+ + e = MnS(s) + 2H_2O$	+527	+407	+287
NO_3^- — NH_4^+	$1/8\ NO_3^- + 5/4\ H^+ + e = 1/8\ NH_4^+ + 3/8\ H_2O$	+444	+369	+294
NO_2^- — NH_4^+	$1/6\ NO_2^- + 4/3\ H^+ + e = 1/6\ NH_4^+ + 1/3\ H_2O$	+430	+349	+268
$FeOOH(s)$ — $FeS(s)$	$FeOOH(s) + HS^-(3x10^{-6}) + 2H^+ + e = FeS(s) + 2H_2O$	+86	-34	-154
$FeOOH(s)$ — $FeCO_3(s)$	$FeOOH(s) + HCO_3^-(10^{-2}) + 2H^+ + e = FeCO_3(s) + 2H_2O$	+80	-40	-160
SO_4^{2-} — $S^0(s)$	$1/6\ SO_4^{2-} + 4/3\ H^+ + e = 1/6\ S^0(s) + 2/3\ H_2O$	-118	-198	-278
SO_4^{2-} — $H_2S(g)$	$1/8\ SO_4^{2-} + 5/4\ H^+ + e = 1/8\ H_2S(g) + 1/2\ H_2O$	-165	-210	-255
SO_4^{2-} — HS^-	$1/8\ SO_4^{2-} + 9/8\ H^+ + e = 1/8\ HS^- + 1/2\ H_2O$	-158	-225	-292
$S^0(s)$ — $H_2S(g)$	$1/2\ S^0(s) + H^+ + e = 1/2\ H_2S(g)$	-185	-246	-307
$CO_2(g)$ — $CH_4(g)$	$1/8\ CO_2(g) + H^+ + e = 1/8\ CH_4(g) + 1/4\ H_2O$	-186	-247	-308
$N_2(g)$ — NH_4^+	$1/6\ N_2(g) + 4/3\ H^+ + e = 1/2\ NH_4^+$	-203	-281	-359
$Fe(OH)_3(s)$ — $FeS(s)$	$Fe(OH)_3(s) + HS^-(3x10^{-6}) + 2H^+ + e = FeS(s) + 3H_2O$	-207	-327	-447
$Fe(OH)_3(s)$ — $FeCO_3(s)$	$Fe(OH)_3(s) + HCO_3^-\ (10^{-2}) + 2H^+ + e = FeCO_3(s) + 3H_2O$	-266	-385	-504

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* Reactions are reversible

TOC, BOD, and COD--The major mechanisms for the migration of organic matter in the soil/water system are bioconversion, surface sorption, and metal-organic complexation (Mang, et al., 1978). Bioconversion can either decrease or increase TOC levels in leachates through microbial synthesis and microbial degradation for both soluble and particulate organics. Organics can be released or assimilated through surface sorption by soil particles, especially clay minerals, hydrous oxides, and noncrystalline materials (Greenland, 1971; Fuller, 1978). Chelation may occur in the soil/water system by specific chelates such as microbial slimes, gums, cell debris, humus, lignin polymers, polysaccharides, proteins, and substances with low molecular weights (aliphatic acids, amino acids, organic phosphates, and volatile acid complexes) (Mang, et al., 1978).

Extensive data on the fate of organic matter, primarily in terms of TOC, BOD, or COD in soil/water systems, indicate that organic matter can be substantially attenuated or unaffected depending on the site specific conditions. Griffin and Shimp (1978) and Fuller (1980), demonstrated that COD and TOC were not significantly retained by any soils tested. Griffin (1978) concluded that in a fresh, young leachate, COD would most likely present a high pollution hazard. Fuller (1978) reported that the COD in soil column effluents was initially 30 to 50 times higher than the COD in the applied leachates. Fuller (1978) concluded that soil organic matter initially contributed to the high COD of the soil solution which was subsequently eluted by the applied leachate. The TOC in soil column effluents rapidly increased to the level of TOC in the applied leachate; when the soil columns were subsequently leached with water, essentially all of the applied TOC was eluted from the soil. Engers (1978) found that in down-gradient ground water, the concentration of organic matter in leachate was reduced significantly as a result of filtration and microbiological conversion. In column studies using a recycled simulated landfill leachate, Soyupak, et al., (1978) reported that the influent COD and TOC was reduced by 90 and 94 percent, respectively, over a 41-cm soil column after 70 days of operation. The results obtained in this study can be approximated by a developed model based on biodegradation phenomena. From limited available data (Griffin, et al., 1976; Fuller, 1978) that indicate that the sorption of leachate organic matter by soil is not extensive, Soyupak, et al. (1978) concluded that the microbial decomposition of organic matter in leachate and soil appears to be a significant attenuating mechanism.

Alkalinity--In general, alkalinity in municipal leachates is due primarily to soluble carbonate species. Dissolved silicates, borates, ammonia, organic bases, sulfides, and phosphates can also contribute to the alkalinity. These noncarbonate concentrations are usually small in comparison to the carbonate species. In the soil/water environment, it is suggested that the alkalinity of leachates is controlled by the dissolution/precipitation of metallic carbonate solids in the soil, such as calcite ($CaCO_3$).

Major Ions--

Concentrations of major cations and anions such as sodium, potassium, calcium, magnesium, chloride, and sulfate are high in the soil/water system because (Mang, et al., 1978):

- The solubilities of simple solids of major ions are relatively high;
- The solubilities of clay minerals may be low enough to reduce the concentrations of major ions. However, the nucleation of clay minerals is extremely slow; and
- No other mechanisms can reduce the soluble levels significantly.

Possible mechanisms for regulating the levels of major ions are solubilization and ion-exchange effects. Solubilization is significant only for calcium and magnesium. Calcite, aragonite, dolomite, and brucite regulate concentrations through solubizization. Other complex solids, due to the kinetic constraint, will not play important roles in precipitation/dissolution reactions (Carrol, 1959; Griffin, et al., 1976; Griffin and Shimp, 1978; Chan, et al., 1979).

Ion exchange is one of the important mechanisms for controlling the migration of major ions. Usually occurring on the surface of clay minerals and colloidal organics, the capacity of ion exchange is dependent upon the crystalline structure of the mineral and the chemical composition of the leaching fluids in contact with the mineral. Ion exchange in these minerals is a reverse chemical reaction which follows the law of mass action and is restricted by the number of exchange sites on the mineral and the strength of the bonding of the exchangeable ions to the mineral surface. The ease of replacement of common soil cations is (Rose, 1966):

Li> Na> K> Mg> Ca> Ba> Al

In two municipal soil/leachate interaction studies, chloride and sodium were found to be relatively unattenuated, while potassium and calcium were moderately attenuated (Griffin, et al., 1976; Roulier, 1977). Dilution can be of greater importance than solubilization and ion exchange for the control of the major ions, especially for sodium, potassium, and chloride ions. Therefore, the porosity of the soil and the flow rate of the leaching fluid may become significant factors in regulating the migration of these ions (Mang, et al., 1978).

Nutrients--

Nitrogen compounds--In most natural water systems nitrogen compounds include organic-nitrogen, ammonia-nitrogen, nitrite-nitrogen, and nitrate-nitrogen. The transformation of these nitrogen compounds is greatly affected by biologically mediated reactions which, in turn, are controlled by the types and population of microorganisms, pH, redox conditions, and dissolved oxygen concentrations as well as the concentrations of the compounds themselves. Other mechanisms contributing to the transport of nitrogen compounds are diffusion, sorption, complexation, and hydrolytic reactions (Mang, et al., 1978).

All reactions carried out by living organisms indirectly require nitrogen. Nitrogen fixation* is a process performed by a few bacteria and blue-green algae. If the soil/leachate system is aerobic, nitrification (mineralization) of organic nitrogen sources occurs readily, producing NO_3-N as an end product. Nitrate-nitrogen is highly mobile, moving readily with the soil solution into the lower vadose zone and ultimately into the ground water (Wilson, 1980). Under anaerobic conditions, nitrification is inhibited and the NH_4-N form predominates (ammonification). The incorporation of NO_3-N or NH_4-N into biological tissue is called immobilization or assimilation. Under anaerobic conditions, nitrogen may be returned to the atmosphere through denitrification. Dentrification occurs quite readily as a result of the activity of heterotrophic bacteria, which convert NO_3-N to gaseous N_2 or NO_2.

Organic matter in leachates can play an important role in the reduction and fixation of nitrogen compounds. Most of the nitrogen in municipal refuse is present as organic nitrogen which is only slowly released by the activity of the biotic community. Thus, the rate of biodegradation of organic matter usually influences the nitrogen flux. Many species of bacteria and fungi are capable of reducing organic nitrogen compounds, nitrate, nitrite, and ammonia; others can oxidize nitrogen compounds. All these reactions are enzyme-catalyzed and highly pH dependent.

Due to its cationic nature, ammonia-nitrogen (chiefly in for form of ammonium) may be adsorbed onto solid materials by ion exchange reactions with mineral or organic matter in the soil material. Adsorption is considered one of the most important mechanisms for controlling movement of ammonia-nitrogen through soils and, depending upon the soil environment, might even override biological action (Preul and Schroepfer, 1968). The nitrate anion is relatively mobile and not retained by ion exchange processes.

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* Conversion of atmospheric nitrogen to soluble nitrate ion.

Under many circumstances, nitrogen compounds can be complexed with soluble metal ions (Sillen and Martell, 1964; Sillen and Martell, 1971). There is little information on organic nitrogen species and the thermodynamics of complex formation. However, from data on the complexation of trace metals with humic substances and amino acids (Lu, 1976), it can be surmised that organic nitrogen can also complex with trace metals to a great extent, depending on the concentration of the metals. This complexation effect can increase nitrogen levels in leachates. The importance of the resulting nitrogen complexes depends upon the types and levels of metals and the formation constants, activity coefficients, and pH levels of the leaching solutions in contact with the soil.

Phosphorus compounds--The transport of phosphorus compounds in the soil/water system involves complicated physical, chemical, and biochemical interactions. The most important mechanisms governing the phosphorus migration in the soil/water system are solubilization, sorption, and biological effects (Mang, et al., 1978).

To understand the importance of solubilization effects, phosphate solids possible in municipal leachates have to be assessed. The major phosphate solids which might control the solubilities of phosphate in leachates are shown in Table 47. Among these solids, $Ca_5OH(PO)_3(s)$ and $AlPO_4(s)$ are most likely to control the soluble phosphates in leachates. The equilibrium solubilities of these two solids can be seen in Figure 44. The minimum solubility of $AlPO_4(s)$ is about 0.01 ppm (as P) at pH 6. At pH 5 it increases to 0.02 ppm (as P). At a pH greater than about 6.8, the solubility controlling solid for phosphate will shift from $AlPO_4(s)$ to $Ca_5OH(PO_4)_3(s)$. At pH 7 and an alkalinity of 1000 ppm (as $CaCO_3(s)$), the solubility of the phosphate specie is about 0.35 ppm (as P). From this relationship, it is likely that phosphate solubility is controlled by $AlPO_4(s)$ when leachate pH is less than neutral. When leachate is recirculated or buffered to 7, $Ca_5OH(PO_4)_3(s)$ may become the predominant solid species to govern the soluble phosphate levels.

Although the free soluble phosphate ion is limited by less soluble solids(s), the complexation ions can enhance the phosphate solubility. For orthophosphate, the most important complexation ions in leachates are magnesium and calcium, which will form $CaHPO_4(aq)$ and $MgHPO_4(aq)$ complexes (calculated by Calscience Research, Inc.).

Phosphate adsorbents also can regulate the soluble levels of phosphate (Fox and Kamprath, 1970; Singer, 1972; Chen, et al., 1973; Patrick and Khalid, 1974; Stumm and Morgan, 1981). Hydrated iron oxide is a strong adsorbent for phosphate, however, it seems of little significance in leachate reducing environments because of the release of phosphate caused by the reduction of ferric phosphate solid to the more soluble ferrous phosphate. In general, high phosphate adsorption by clays is favored by a lower pH. Maximum sorption of orthophosphates on montmorillonite occurs at pH 5-6. Sorption by kaolinite is maximum at a pH near 3 (Stumm and Morgan, 1981). It is speculated that phosphate clay adsorbents will not play a important role in landfills when pH levels are close to neutral.

TABLE 47. MAJOR PHOSPHATE CONTROLLING SOLIDS (Strumm and Morgan, 1981)

Equilibrium	Log Equilibrium Constant
$Ca_5OH(PO_4)_3(s) = 5Ca^{2+} + 3PO_4^{3-} + OH^-$	-55.6
$Ca_5OH(PO_4)_3(s) + 3H_2O =$ $2[Ca_2HPO_4(OH_2)_2]$ surface $+ Ca^{2+}$ $+ HPO_4^{=}$	-8.5
$[Ca_2HPO_4(OH)_2]_{\text{surface}} = 2Ca^{2+} + HPO_4^{2-} + 2OH^-$	-27
$CaHPO_4(s) = Ca^{2+} + HPO_4^{2-}$	-7
$FePO_4(s) = Fe^{3+} + PO_4^{3-}$	-23
$AlPO_4(s) = Al^{3+} + PO_4^{3-}$	-21
$Ca_2P_2O_7(s) = Ca^{2+} + P_2O_7^{2-}$	-7.9

In view of the above, the soluble phosphate levels in the neutral leachates are most likely controlled by $Ca_5OH(PO_4)_3(s)$ and regulated by complexation ions. Because of this phenomenon, the calcium ion concentration or alkalinity level in leachates will be the dominant factor affecting the soluble phosphate levels. The concentrations of the major soluble phosphate (mainly orthophosphate) species were calculated and are shown in Figure 45.

As can be seen from Figure 45, the total soluble phosphate levels in leachates may range from 0.30 to 6.5 ppm (as P) when pH ranges from 5.5 to 7.0 and when alkalinity ranges from 1000 to 6000 ppm (as $CaCO_3$). High alkalinity levels favor the

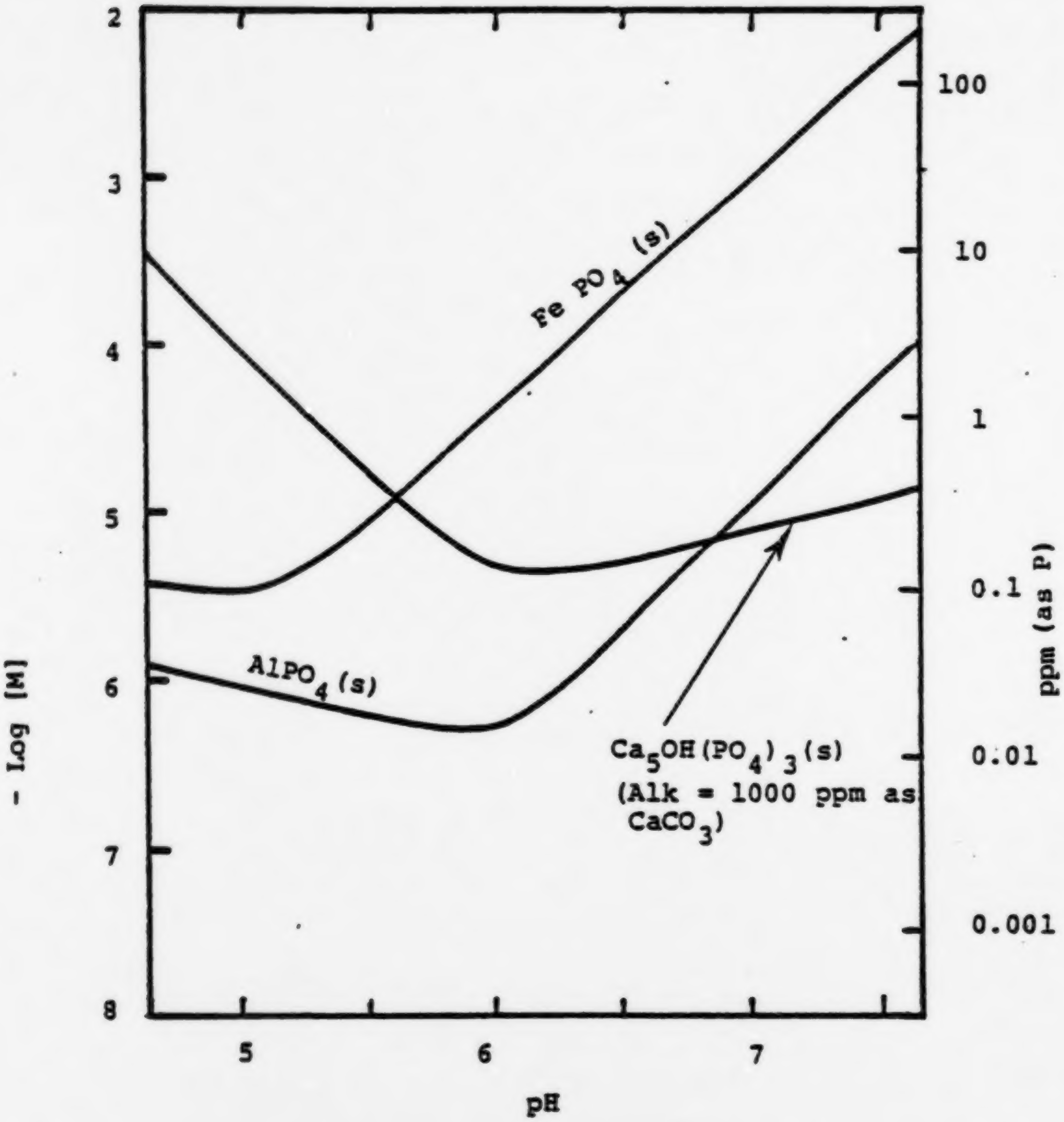
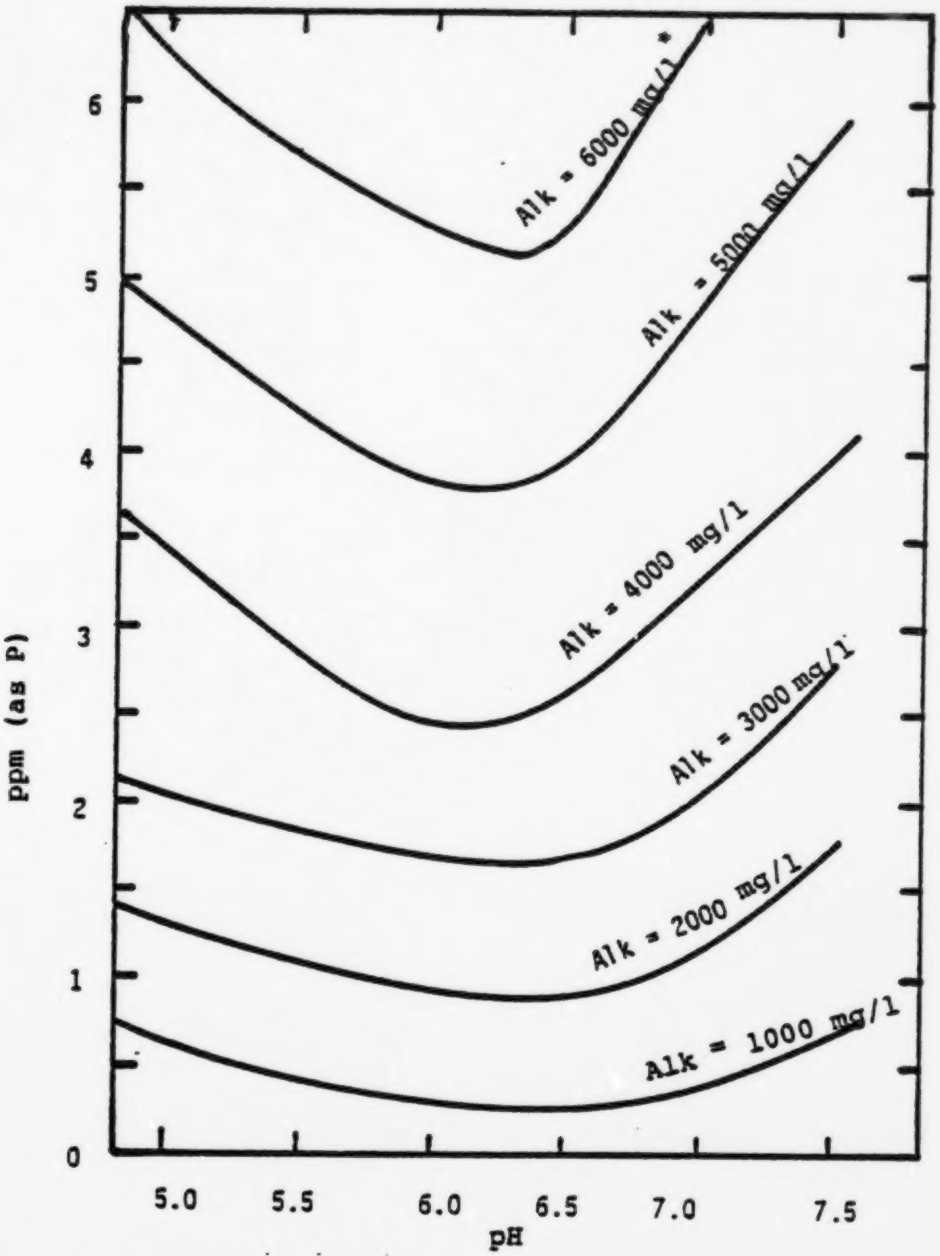


Figure 44. Solubilities of major phosphate solids (after Strumm and Morgan, 1981).



* mg/l (as $CaCO_3$)

Figure 45. Total soluble phosphate levels in leachates as affected by pH and alkalinity.

solubilization of phosphate in leachates. At the same alkalinity, pH values between 6.0 to 6.5 appear to be least soluble for phosphate, and when the pH increases or decreases from this range, the soluble phosphate levels exhibit an increasing trend (see Figure 45).

Trace Metals--

The movement of trace metals in the soil/water system is extremely complex. Not only are there numerous controlling factors, but many unknowns beyond present state-of-the-art knowledge are involved. Thermodynamic and kinetic influences make transport phenomena difficult to explain and predict. Some of the major mechanisms that influence mobility of trace metals in solid-liquid interfaces are:

- Solubilization;
- Sorption;
- Complexation; and
- Dilution.

Each of these mechanisms may exert some influence on the migration of trace metals in soil/water systems.

The importance of solubilization effect for trace metals has been discussed previously. Griffin, et al., (1976) suggested that migration of Pb, Zn, Cd, and Hg in the soil/water system could be explained by the solubilization effect. It also has been suggested by Mang, et al.,(1978) and Lu (1981) that the concentrations for most trace metals in leachates could gradually approach the equilibrium solubility levels of predominant solids after long-term contact of leachate and media. Although the solubility effect is an important factor for governing trace metal migration, it is usually suggested that this mechanism is significant only in alkaline or reducing environments (Lu, 1976; Mang, et al., 1978; Duvel, et al. 1979; Chen, et al., 1980; Lu 1981).

In a more acidic environment, adsorption mechanisms were suggested to be more important for regulating the levels of trace metals in leachates. Content of hydrous oxides (primarily iron, manganese and aluminum hydrous oxides), clay minerals, pH, soil and leachate organic species, and specific competing ions (especially major cations) were concluded to be the most significant factors affecting trace metal adsorption (Gadde and Laitinen, 1974; Leckie and James, 1974; Korte, et al., 1975; Griffin, et al., 1976; Fuller, 1977; Roulier, 1977; Fuller, 1980; Griffin and Shimp, 1978). Due to a wide variety of soils and leachates, development of a quantitative generic model to explain the adsorption of trace metals in the soil/water system is still beyond our present knowledge.

Metallic complex formation is another important factor controlling the total concentrations of trace metals in the soil/water system (Schnitzer, 1971; Leckie and James, 1974; Weber and Possett, 1974; Lu, 1976; Stevenson, 1976; Mang, 1978; Stumm and Morgan, 1981). In a system containing only inorganic species, the quantification of the complexation effect could usually be estimated successfully (Lu, 1980). Because of the many unknown organic species and lack of knowledge concerning thermodynamic data in the soil/water system, a precise evaluation of all the metallic complexes is virtually impossible.

The difference in the rates of the aforementioned reactions and the dilution effect (controlled by the solution flux of both leachate and ground water) also affects the transport of trace metals in the soil/water system. The leachate flow rate is proportional to the moisture content of the soil through which leachate is being transported. Dilution by ground water may be important in some cases; however, the availability of ground water for dilution may be limited by the hydrogeography of the site. The presence of clay or sand lenses, rock fissures, etc., can limit drastically the water available for dilution, and can transport leachates long distances without allowing dilution to take place (Ham, 1975).

Chlorinated Hydrocarbons--

It is difficult to predict the fate of chlorinated hydrocarbons in the environment. In general, volatilization, microbial degradation, chemical hydrolysis, oxidation, and sorption can be involved (Boucher and Lee, 1972; Nisbet and Sarofin, 1972; Davison, 1978; Farmer, 1978; Griffin, 1978; Griffin and Chou, 1980). These studies suggest that sorption may be the dominant mechanism in soil/water systems.

Previous studies have shown that chlorinated hydrocarbons tend to be strongly sorbed by soils. The major adsorbents are clay minerals, iron and manganese hydrated oxides, and organic material (Davison, 1978; Griffin, 1978; Griffin and Chou, 1980). The rate of adsorption of chlorinated hydrocarbons studies was found to be rapid, with equilibrium conditions achieved within a few hours (Griffin, 1978). The adsorption process conformed to the Freundlich adsorption equation (Davison, 1978; Griffin, 1978). In general, chlorinated hydrocarbons are highly resistant to aqueous phase mobility through earth materials, because of their low water solubilities and strong sorption by soils; however, they are highly mobile in organic solvents (McCarty and Rittman, 1981).

Pesticides--

Factors influencing the fate of pesticides in soil/water systems include: (1) adsorption/desorption, (2) microbial decomposition, (3) volatilization, (4) soil moisture, and (5) physical properties of the soil (Baily and White, 1970; Leonard, et al., 1976). Adsorption/desorption is considered to be the prime factor governing the interactions between pesticides and soil colloids.

Organic pesticides were classified as ionic or nonionic by Leonard, et al. (1976). Ionic pesticides are subclassified as cationic (paraquat, disquat), basic (S-triazones), and acidic (benzoic acids, phenolds, picolinic acid). Nonionic pesticides include chlorinated hydrocarbons and organophosphates. Cationic pesticides are strongly retained on the exchange complex. Changes in soil pH have a complex effect on pesticides forms. For example, Leonard, et al., 1976, reported that a decrease in pH increases the molecular form of an acidic pesticides but increases the conjugate acid form of a basic pesticides. These changes will modify adsorption/desorption characteristics and the mobility of pesticides. Leonard, et al. (1976) also reviewed the effects of clay content and soil moisture on adsorption. It was reported: (1) adsorption is enhanced with increased organic content, and (2) a decrease in soil moisture may increase surface acidity, and, thus, increase adsorption.

Microbial decomposition of pesticides depends on such factors as microbial population, soil moisture and temperature, organic matter content, pH, redox potential, pesticide concentration, availability for degradation, and nutrient concentration and availability. Leonard, et al. (1976) presents a detailed discussion of these factors and their effects on microbial decomposition of pesticides.

Viruses--

Chang and Page (1979) cautioned that immobilizaton of viruses by soil should not be equated with virus inactivation because many adsorbed virus particles have been demonstrated to be infectious for significant periods of time. Viruses immobilized by soil adsorption may also become desorbed when the chemical composition or pH of the percolating wastewater is changed.

Recent studies by Keswick and Gerba (1980) indicate that virus survival within a soil relate to soil moisture content, temperature, pH, nutrient availability, and antagonisms. An evaluation of these factors indicates that because of the detrimental effects of aerobic soil microorganisms, evaporation, and higher temperatures, inactivation of viruses will be much more rapid near the surface than for those viruses that penetrate the soil more deeply. Thus, viruses which penetrate the soil surface are expected to survive for prolonged periods of time, as compared to those retained near the soil surface.

Laboratory studies have shown that viruses have widely different adsorption properties (Wellings, 1974; Green and Oliver, 1975). This is, therefore, an important factor in the movement of viruses from sources of contamination into otherwise clean ground water. Thus, for evaluation of a site, several types of viruses should be tested in order to obtain a representative picture (Keswick and Gerber, 1980).

The studies by Wellings (1974, 1975) were among the first to demonstrate transport of viruses for long distances through soils into ground water. Specific factors affecting the migration of viruses in soils are summarized in Table 48.

TABLE 48. FACTORS THAT INFLUENCE THE MOVEMENT OF VIRUSES IN SOILS (after Chang and Page, 1979; Keswick and Gerba, 1980; Wilson, 1980).

Factor	Remarks
Soil Composition	Clay soils are good adsorbers of viruses under appropriate conditions. Sandy loam soils and organic soils are also favorable for virus removal. Soils with a low specific surface area are most effective for virus removal.
pH	Virus removal increases with decreasing pH; high pH results in elution of adsorbed virus.
Soluble Organics	Soluble organic matter competes with viruses for adsorption sites on the soil particles, resulting in decreased virus adsorption or elution of an already adsorbed virus. Definitive information is not available for soil systems.
Cations	The presence of cations usually enhances the retention of viruses by soil.
Rainfall	Viruses near the soil surface may be eluted after a heavy rainfall. Evapotranspiration will tend to increase the concentration of viruses at the soil surface.

TYPES OF LEACHATE MIGRATION MODELS

Whenever liquid or solid wastes are deposited on land, leachates may be generated which can seriously degrade the quality of underlying ground-water systems. Predicting the potential magnitude of such ground-water pollution is a complex technological undertaking. The simultaneous presence of numerous physical, chemical, and biological interactive mechanisms makes it very difficult to obtain an understanding of the pollution potential of a given waste for a specific geohydrological environment. Consequently, many designers/investigators have resorted to the use of simulation models to obtain information necessary for rational landfill design and to forecast the performance of a specific waste disposal site.

A waste disposal model represents a simplified delineation of a real waste disposal system. As a result of simplification, different types of models exist. Leachate migration models may be classified according to a number of schemes. One such arrangement provides the four following generic model categories:

- Descriptive models;
- Physical models;
- Analog models; and
- Mathematical models.

Table 49 lists a few example models and their classification into different groupings. Each of these model categories are described in the following text.

Descriptive Models

A descriptive model utilizes a qualitative judgment approach to evaluate a landfill site rather than a well defined, quantified set of steps. While the cost of this approach is a favorable consideration, the highly subjective nature of the method severely limits its validity because different experts may reach different conclusions based upon this modeling approach. Because of these subjective, site-specific, and qualitative limitations, descriptive models are not further addressed in this report.

Physical Models

A physical model is a scaled-down version of the actual landfill condition. Only a few physical landfill models have been constructed in the past (Quasim, 1965; Fungaroli and Steiner, 1973; Pohland, 1975). Although physical models of

TABLE 49. EXAMPLE MODELS AND THEIR CLASSIFICATION INTO DIFFERENT GROUPS (van Genuchten, 1978)

MODEL DEFINITION	TYPE OF MODEL				
On site inspection using engineering judgement	Descriptive	Empirical	Deterministic	Dynamic	3D*
The Drexel University experimental field landfill (Fungaroli and Steiner, 1973)	Physical	Empirical	Deterministic	Dynamic	3D
Batch equilibrium study; shaker test; solid waste evaluation leachate test (subsystem models)	Mathematical	Empirical	Deterministic	Static	0D
Column displacement studies; thin-layer chromatography (subsystem models)	Mathematical	Empirical	Deterministic	Dynamic	1D
Criteria listing; matrix method	Mathematical	Empirical	Deterministic	Static	0D
One-dimensional unsaturated transport model of Bresler (1973) (subsystem model)	Mathematical	Conceptual	Deterministic	Dynamic	1D
Two-dimensional saturated-unsaturated transport model of Duguid and Reeves (1976)	Mathematical	Conceptual	Deterministic	Dynamic	2D
Model for ground-water flow and mass transport under uncertainty of Tang and Pinder (1977)	Mathematical	Conceptual	Stochastic	Dynamic	2D

* Indicates spatial dimension: 3D - three-dimensional, etc.

waste disposal sites are generally lacking, experiments can be conducted to aid investigators in making more accurate predictions. Such experimentation may include column leaching studies, thin-layer chromatography, or batch equilibrium studies. Unfortunately, this information does not, in itself, define a waste disposal model; therefore, it cannot be used as a predictive tool. However, experimental data such as adsorption and decay constants, dispersion coefficients, etc. may provide necessary information for development of a mathematical model. While it is obvious that scaled-down physical models can provide useful information, their practical application as a predictive tool appears doubtful. They are costly to build and also may be very time-consuming to study, especially in consideration of the different chemical and biological processes within a landfill which may be operative over a period of many years.

Analog Models

Analog models employ a convenient transformation of a given physical system into another one which behaves in a similar manner, but which is measured more easily. Electrical analog models have found application in ground-water flow modeling (Jorgenson, 1975). Because of the high cost of building electrical analog models for large-scale field problems, it is doubtful that many such models will be used in the near future for the simulation of ground-water quality problems.

Mathematical Models

Mathematical models employ a set of concise mathematical equations to describe the relationships between the various systems parameters and their input and output variables. The series of variables usually represent known physical and chemical properties, and relationships, which can be solved using either analytical or numerical techniques. A compilation of different types of mathematical models is given in Table 50.

MODEL DISTINCTIONS

Several distinctions between models can be made, depending upon the method of analysis defined by the model and the particular approach used to solve the model. Common model distinctions are as follows:

- Empirical or conceptual models;
- Deterministic or stochastic models;
- Static or dynamic models; and
- Spatial dimensionality of the model (i.e., 1, 2, or

TABLE 5 6. PARTIAL LIST OF AVAILABLE TRANSPORT MODELS FOR APPLICATION TO GROUND-WATER QUALITY PROBLEMS

Model No.	Model References	Geometry of Model[1)]	Method of Solution[2)]	Type of Flow[3)]	Type of Soil[4)]	Type of Chemical Inter- actions[5)]	Application/Comments
A.	SATURATED-UNSATURATED TRANSPORT MODELS						
A1	Duguid and Reeves (1976, 1977)	2D, C	LFE	Tr	L, An	Ad, De	Transport of radionuclides from a waste-disposal site.
A2	Segol (1976, 1977)	2D, 3D	HFE	Tr	L, An	--	--
A3	van Genuchten *et al*. (1977)	2D, C	HFE	Tr	L, An	Ad, De	Leachate movement from a hypothetical landfill.
A4	Sykes (1975)	2D, C	HFE	St	L, An	--	Contaminant movement from a landfill.
A5	Elzy *et al*. (1974)	2D, C	O	Tr	--	Ad, De	Contaminant movement from a landfill.
A6	Perez *et al*. (1974)	2D, C	FD	Tr	L	--	Groundwater pollution from agricultural sources.
A7	Shaffer *et al*. (1977)	2D, C	FD	Tr	L	Ad, Ce, De	Irrigation return flow model; coupled 1D, vertical unsaturated model, and 2D, saturated model.
B.	SATURATED-ONLY TRANSPORT						
B1	Gupta *et al*. (1975)	3D	HFE	Tr	L, An	--	Simulates rising connate water through a vertical fault in multi-aquifer system (steady state flow application only).

(Continued)

TABLE 50.(Continued)

Model No.	Model References	Geometry of Model[1]	Method of Solution[2]	Type of Flow[3]	Type of Soil[4]	Type of Chemical Interactions[5]	Application/Comments
B2	Gureghian and Cleary (1977)	3D	LFE	St,	An	Ad, De	Applied to an existing landfill on Long Island.
B3	Pickens and Lennox (1976)	2D, C	TFE	St.	L, An	Ad	Contaminant transport from a hypothetical landfill.
B4	Schwartz (1977)	2D, C	MOC	St	An	Ad, Co, De	Hypothetical study of subsurface pollution by radioactive wastes (1975); model analysis of a proposed waste-management site (1977).
B5	Bredehoeft and Pinder (1973)	2D, A	MOC	Tr	An	--	Movement of salt water in confined limestone aquifer; predicted future concentrations and tested effects of protective pumping.
B6	Konikow and Bredehoeft (1974)	2D, A	MOC	Tr	An	Ad, De	Used calibrated model to evaluate effects of different irrigation practices on salinity changes in an alluvial stream-aquifer system.
B7	Robertson (1974), Robertson and Barraclough (1973)	2D, A	MOC	Tr	An	Ad, De	Transport of industrial and low-level radioactive wastes into the Snake River Plain aquifer, Idaho. Simulated 20 year history of pollution.
B8	Robson (1974)	2D, A	MOC				Pollution of shallow aquifer by seepage from sewage treatment ponds; predicted future concentrations and tested alternative water management plans.
B9	Robertson (1974)	2/3D	A/MOC	Tr	An, L	Ad, De	Three-segment model for flow, including ability to simulate perched water in the unsaturated zone (see also B7).
B10	Konikow (1977)	2D, A	MOC	St			Simulated 30 year history of groundwater pollution by chloride from an unlined disposal pond into the underlying alluvial aquifer.
B11	Helweg and Labadie (1976)	2D, A	MOC	Tr	An	--	Adapted version of B6; used as a cost-effective salinity management technique for stream-aquifer systems.

(Continued)

TABLE 5 6. (Continued)

Model No.	Model References	Geometry of Model 1)	Method of Solution 2)	Type of Flow 3)	Type of Soil 4)	Type of Chemical Interaction 5)	Applications/Comments
B12	Grove (1970)	2D	FE			Ad, De	Transport of industrial and low-level radioactive wastes into interbedded basalt flows and unconsolidated sediments.
B13	Reddell and Sunada (1970)	2D, A, C	MOC	Tr	An	Ad, Ce	Three-dimensional formulation, two-dimensional application only.
B14	Ahlstrom and Baca (1974)	2D, A	MOC	St/Tr	An	Ad, Ce	Considers adsorption and exchange of several macro- and micro-ions.
B15	Pinder (1973)	2D, A	UFE	St	An	--	Described and predicted future concentrations of hexavalent chromium seeping from a waste disposal pit into underlying glacial outwash aquifer.
B16	Thomas *et al*. (1977; Martinez *et al*. (1975)	2D, A	UFE	St	--	Ad, De	Describes ground-water pollution from salt dome leachates.
B17	Besbes *et al*. (1976)	2/3D, A	FD	Tr	L	--	Areal model for multilayered aquifersystem; predicted concentration changes after dam construction in the Kairouan Plain, Tunesia.
B18	Fried (1975) Fried and Ungemach (1971)	2D, A	FD	Tr	--	--	Flow part based on Boussinesq equation; describes pollution by NaCl from large salt dumps into alluvial aquifer in Northeastern France.
B19	Lessi (1976)	2D, C	UFE	Tr	L	--	Applied to solute transport in a heterogeneous aquifer.
	SALTWATER INTRUSION MODELS						
B20	Pinder and Page (1977)	2D, A	TFE	Tr	--	--	Vertically integrated sharp-interface salt water intrusion model. No transport equation is solved.
B21	Segol, Pinder and Gray (1975)	2D, C	UFE	St	An	--	Calculating the position of the saltwater front.
B22	Lee and Cheng (1974)	2D	MOC	Tr	An	--	Seawater encroachment in coastal aquifers.
B23	Green and Cox (1966)	2D	MOC			--	Storage of fresh water in underground reservoirs containing saline water.
B24	Pinder and Cooper (1970)	2D, C	MOC	St	--	--	Calculating the transient position of a saltwater front.

(Continued)

TABLE 5 6. (Continued)

Model No.	Model Reference	Geometry of Model[1]	Method of Solution[2]	Type of Flow[3]	Type of Soil[4]	Type of Chemical Interactions[5]	Applications/Comments
C	UNSATURATED-ONLY TRANSPORT MODELS						
C1	Bresler (1975)	2D, C	FD	Tr	--	--	Describes two-dimensional transport of solutes under a trickle source.
C2	Hildebrand and Himmelbau (1977); Hildebrand (1975)	1D	FD	--	--	--	Transport of nitrate in a sand column.
C3	Bresler (1973)	1D	FD	Tr	--	--	Compared results with field data on chloride transport during infiltration.
C4	Wood and Davidson (1975); Davidson et al. (1975).	1D	FD	Tr	--	Ad	Applied to pesticide transport.
C5	Ungs et al. (1976)	1D	FD	Tr	--	Ad, De	Compared results with observed field data on chloride transport during infiltration.
C6	Selim et al. (1976)	1D	FD	Tr	--	Ad	Applied to transport of 2,4-D in soils.
C7	Shah et al. (1975)	1D	FD	St	L	Ad	Applied to phosphorus transport in soils; assumes constant dispersion coefficient. and kinetic model for phorphorus adsorption.
C8	Kirda et al. (1973)	1D	FD	Tr	--		Applied to anion movement in soil columns
C9	Tanji et al. (1967) Tanji et al. (1972)	1D	FD	St	L	Ad, Ce	Approximate solutions for cation exchange in field soils.
C10	Dutt et al. (1972)	1D	FD	Tr	L	Ad, Ce	
C11	Rubin and James (1973)	1D	LFM	St	--	Ad, Ce	
C12	van Genuchten and Pinder (1977)	1D	HFE/FD	Tr	L	Ad, De, Ce	Modeling of leachate and soil interactions in an aquifer.
C13	Gureghian et al. (1977)	1D	FD	Tr	L	Ad, De, Ce	Simulation of pollutant transport in Long Island, N.Y.
C14	King and Hanks (1973, 1975)	1D	FD	Tr	--	Ad, De, Ce	Applied to irrigation return flow quality studies. Includes plant root uptake of water.
C15	Gaudet et al. (1977)	1D	FD	St	--	--	Applied to soil with mobile and immobile water.

(Continued)

TABLE 5 0.(Concluded)

Model No.	Model Reference	Geometry of Model[1)]	Method of Solution[2)]	Type of Flow[3)]	Type of Soil[4)]	Type of Chemcial Interactions	Applications/Comments
C16	Selim, et al.(1977)	1D	HFE/FD	Tr	L	Ad	Modeling of leachate and soil interactions in an aquifer.
C17	Warrich, et al. (1971)	1D	FD/A	Tr	--	--	Approximate analytical solution of transport equation; applied to field irrigation study with chloride.
C18	Smajstrala, et al. (1975)	1D	MOC	Tr	--	--	Miscible misplacement in soils.
D	ANALYTICAL TRANSPORT MODELS						
D1	Kuo (1976), Shen (1976) Cleary, et al. (1973) Wang, et al. (1977) Yeh and Tsai (1976), Cleary (1976)	2,3D	A	St	--	(Ad, De)	Various applications and assumptions
D2	Lapidus and Amudson (1952), Brenner (1962) Lindstrom, et al.,(1967) Lindstrom and Boersma (1971, 1973), Lindstrom and Stone (1974a, b), Cleary and Adrian (1973) Marino (1974) Ogata (1961), van Genuchten and Wierenga (1976), Selim and Mansell (1976), Soyupak, et al. (1978), Fuller, (1978), Frere, (1980) Leonard and Wauchope (1980), Davidson (1980)	1D	A	St		(Ad, De)	Various applications, including: - zero and first order decay; - linear equilibrium adsorption; - first order kinetic adsorption; - solute transfer between mobile and immobile water; - decay boundary conditions.

1)
1D = one-dimensional
2D = two-dimensional
3D - three dimensional
A = Areal (2D only)
C = Crossectional (2D only)

2)
A = Analytical
FD = Finite difference
LFE = Linear finite elements
TFE = Triangular finite elements
HFE = Mixed/higher order finite elements

MOC = Method of characteristics
O = Other

3)
Tr = Transient
St = Steady-state

4)
L = Layered
An = Anisotropic

5)
Ad = Adsorption
Ce = Cation exchange (multi-ion transport)
De = Decay

3 dimensions).

These additional classification schemes are described in the following text.

Empirical or Conceptual Models

The delineation between an empirical and conceptual model is dependent upon whether or not the assumed physical processes use input variables to produce output variables. Empirical models are based completely upon observation and/or experimentation, while conceptual models use equations based upon the conservation of mass, energy, and momentum. Examples of these models are presented in Table 49.

Deterministic or Stochastic Models

In a deterministic model, all input variables and system parameters are assumed to have fixed mathematical or a logical relationship with each other. These relationships completely define the system (a unique solution is obtained). See Table 49 for examples of these models.

Stochastic (or probabilistic) models, on the other hand, take into account the intrinsic randomness associated with any of the system parameters, or the uncertainties associated with the many mechanisms operating in the system, the system parameters themselves, or the input variables. Within the stochastic model category, two types exist. These are:

1. Stochastic models where the system parameters and input variables are characterized by assumed probability distributions (normal, log-normal, etc.) which are used to characterize the statistical behavior of selected parameters (Freeze, 1975).

2. Stochastic models where the system parameters or input variables are uncertain either because of unreliable input data, or due to measurement errors. Uncertainty may also result from the case of an over-simplified model where different mechanisms are often bunched together, resulting in less well-defined parameters. Tang and Pinder (1977) used a mean and variance approach to describe the uncertainty of applicable parameters. Stochastic models generally determine a confidence interval for each of the ouput variables.

Static or Dynamic Model

Static models evaluate steady-state conditions, i.e., where the input and ouput variables do not change with time.

Conversely, a model whose input and output variables change with time are termed dynamic or transient (see Table 49).

Spatial Dimensionality of the Model

The spatial dimensionality of the model refers to whether it considers one, two, or three dimensions. A one-dimensional model, for example, could be used to describe the vertical transport of contaminants in the unsaturated zone between the landfill and the ground-water table. Such a model, however, will not provide accurate information regarding ground-water pollution because the effects of leachate dilution by ground water cannot be evaluated. In general, however, at least a two-dimensional cross-section model must be developed in order to obtain the desired information. Two-dimensional models may also be formulated on an areal basis whereby the system parameters and input and output variables represent average quantities along the vertical dimensions. Three-dimensional models are utilized to obtain a more representative characterization of an actual setting. Examples of one-, two-, and three-dimensional models are given in Table 50.

CONCEPTUAL-MATHEMATICAL MODELS

Of the different types of models previously discussed, conceptual-mathematical models appear to be the most promising, but unfortunately, also the most complex ones for evaluating potential ground-water pollution from waste disposal sites (van Genuchten, 1978). A compilation of the different types of conceptual-mathematical models available for possible use in ground-water quality evaluation studies is given in Table 50. This compilation by Weston, Inc. (1978) is based largely on contributory material prepared by van Genuchten (1978). The list of models in Table 50 is not intended to be complete; its purpose is to demonstrate the existence of a wide variety of models, to characterize their most important capabilities and limitations, to identify their method of solution, and to show some of their applications. The list of conceptual-mathematical models in Table 50 is delineated according to the following categories;

- Saturated and unsaturated transport models;
- Unsaturated only transport models;
- Saturated only models; and
- Analytical transport models.

Each type is described briefly in the following text.

Saturated - Unsaturated Transport Models

Saturated - unsaturated transport models simulate either a three-dimensional system (Model A2) or a two-dimensional cross section. Except for Model A7, no cation exchange (multi-ion transport) is considered in any of the models in this group, though several include single-ion adsorption and decay (Models A1, A3, and A5).

Saturated - unsaturated transport models are probably the most appropriate ones for landfill simulation because they consider the unsaturated conditions in and under the waste disposal site. Another application of these models is the determination of the dilution of leachate (an important attenuation mechanism) by flowing ground water.

Saturated-Only Transport Models

These models deal exclusively with saturated flow and ignore the dynamics of the unsaturated zone between the waste disposal site and the ground-water table. This approach is most conducive to contaminant transport when the ground water saturates the waste material. Many saturated-only models use the method of characteristics (MOC) for solving the governing transport equation. Most of these models are adaptations of mass flow approaches for fluid flow.

A special class of saturated-only transport models is provided by salt water intrusion studies (Models B20-B24). These models differ from the other (cross-sectional) models in this group in that they consider density-dependent flow, and, as such, are applicable to contaminant transport from water disposal sites.

Unsaturated-Only Transport Models

These one-dimensional models are useful for studying the mechanisms of contaminant transport in the unsaturated zone. Another important application of these models is obtained when they are used simultaneously with saturated-only transport models. The unsaturated models can then be used to predict both the amount and type of leachate reaching the ground-water table, information which can be used as input for the saturated-only transport models. Models A6 and A7 represent examples of this type of approach.

Analytical Models

Two- and three-dimensional analytical models appear to have limited application to actual field ground-water contamination problems. Their application is generally restricted to one-dimensional models based on constant porosity, dispersivity, and conductivity. The different one-dimensional analytical models (Models D2 in Table 50) are potentially useful tools for identification and quantification of waste-soil interactions when used in conjunction with column leaching studies.

Conceptual-mathematical models are based upon a set of equations which describe relationships between different input and output variables as well as system variables. After suitable simplifications, the equations generally reduce to a set of coupled non-linear, second-order partial differential equations. Of these partial differential equations, one will describe fluid flow while the others characterize the transport and behavior of different chemical constituents associated with the waste leachate.

Numerous conceptual-mathematical models are presently available with differences between the models arising from the numbers of simplifications made during the derivation of the basic governing equations, the method of solving the equations, and the stipulated boundary conditions.

The following partial differentail equations may be used to simulate the (density independent) transport of water and dissolved constituents in a saturated-unsaturated three-dimensional medium, such as occurs in a landfill and the underlying groundwater system (van Genuchten, 1978).

Mass transport equation:

$$\underset{(a)}{\rho \frac{\partial S_k}{\partial t}} + \underset{(b)}{\frac{\partial \theta c_k}{\partial t}} = \underset{(c)}{\frac{\partial}{\partial x_i} \left(\theta D_{ij} \frac{\partial c_k}{\partial x_j}\right)}$$

$$\underset{(d)}{- \frac{\partial}{\partial x_i} (q_i c_k)} \pm \underset{(e)}{\sum_{m=1}^{\ell_1} (a_m \theta c_k^m)}$$

$$\underset{(f)}{\pm \sum_{m=1}^{\ell_2} R_k} \underset{(g)}{\pm Q c_k^*} \text{ --- } (39)$$

Flow equation:

$$\left(\frac{\theta}{n} S_s + C\right) \frac{\partial h}{\partial t} = \frac{\partial}{\partial x_i} \left(K_{ij} \frac{\partial h}{\partial x_j} + K_{i3}\right) \pm Q$$

$$(\ i,\ j = 1,2,3) \text{ ---------------------------- } (40)$$

Table 51 provides an explanation of the symbols used in these equations. Processes described in this constituent transport equation are as follows:

- Changes in adsorbed constituent concentration;
- Changes in solution constituent concentration;
- Diffusion/dispersion effects;
- Convective transport of the constituent by the fluid;
- Production (+) or decay (-) reactions;
- Additional chemical/soil or chemical/chemical interactions (precipitation, chemical transformations, cation exchange reactions, volatilization, etc.); and
- Constituent concentration changes resulting from water sources (+) or sinks (-).

The volumetric fluid velocity, q_i, is required before the transport equation can be solved. For this, it is necessary to also solve the water flow equation. The water flow equation may be solved to provde either a steady state flow simulation ($\frac{\partial h}{\partial t} = 0$), or repeatedly, in the case of a transient simulation. Whatever approach is used, the volumetric fluid velocity, q_i, is obtained from Darcy's Law (Schwab, et al., 1966) in the form of:

$$q_i = - K_{ij} \frac{\partial h}{\partial x_j} - K_{i3} \text{ ------------------------------} \quad (41)$$

The transport equation is also coupled to the flow equation through the dispersion coefficient, D_{ij}. The magnitude of D_{ij} depends upon the fluid flow velocity, q_i, and the soil moisture content, Θ (which in turn is determined from the pressure head).

The constituent transport equation allows the incorporation of an adsorption component describing the dependency of the sorbed constituent concentration, S, on the solution concentration, c, through the use of appropriate adsorption isotherms. Models available for adsorption and/or ion exchange are classified into either equilibrium models which assume instantaneous chemical adsorption, and kinetic models which consider the approach towards equilibrium. Table 52 lists a few of the most frequently used equilibrium and kinetic absorption models. Most of the equilibrium models in Table 52 represent special cases of kinetic models, in that, they are derived by setting the time derivative $\frac{\partial s}{\partial t}$, equal to zero.

TABLE 51. EXPLANATION OF SYMBOLS USED IN THE MASS TRANSPORT AND FLOW EQUATIONS

Symbol	Explanation
c_k	Solution concentration of chemical species k (ML^{-3});
c_k^*	Solution concentration of the fluid source or sink (ML^{-3});
C	Specific soil moisture capacity, $C = n \frac{\partial s_w}{\partial h}$ (L^{-1});
D_{ij}	Dispersion coefficient (L^2T^{-1});
h	Pressure head (L);
K_{ij}	Soil hydraulic conductivity (LT^{-1});
n	Porosity (L^0);
q_i	Volumetric fluid velocity (LT^{-1});
Q	Fluid source or sink term, $Q = Q_w \delta(x_i - x_{wi})$ (T^{-1});
Q_w	Strength of fluid source or sink (L^3T^{-1});
R_k	Rate term expressing soil-chemical and chemical-chemical interactions ($ML^{-3}T^{-1}$);
s_k	Adsorbed concentration of chemical species k (M^0);
S_s	Specific storage coefficient (L^{-1});
S_w	Degree of fluid saturation (L^0);
t	Time (T);
x_i	Distance in i-th coordinate direction (L);
x_{wi}	i-th coordination of source or sink (L);
α_m	m-th order rate constant for production or decay ($M^{1-m}L^{3m-3}T^{-1}$);
δ	Dirac delta function (L^{-1});
ρ	(Dry) soil bulk density (ML^{-3});
θ	Volumetric moisture content (L^0).

TABLE 52. PARTIAL LIST OF EQUATIONS USED TO DESCRIBE ADSORPTION REACTIONS*

MODEL	EQUATION	REFERENCE
1. Equilibrium		
1.1 (linear)	$s = k_1 c + k_2$	Lapidus and Amundson (1952) Lindstrom, *et al.* (1967)
1.2 (Langmuir)	$s = \frac{k_1 c}{1 + k_2 c}$	Tanji (1970) Ballaux and Peaslee (1975)
1.3 (Freundlich)	$s = k_1 c^{k_2}$	Lindstrom and Boersma (1970) Swanson and Dutt (1973)
1.4	$s = k_1 c e^{-k_2 s}$	Lindstrom, *et al.* (1971) van Genuchten, *et al.* (1974)
1.5 (Modified Kjelland)	$s = \frac{c s_m}{c + k_1 (1-c) \exp [k_2 (c_m - 2c]}$	Lai and Jurinak (1971)
2. Non-equilibrium		
2.1 (linear)	$\frac{\partial s}{\partial t} = k_r (k_1 c + k_2 - s)$	Lapidus and Amundson (1952) Oddson, *et al.* (1970)
2.2 (Langmuir)	$\frac{\partial s}{\partial t} = k_r (\frac{k_1 c}{1 + k_2 c} - s)$	Hendricks, 1972
2.3 (Freundlich)	$\frac{\partial s}{\partial t} = k_r (k_1 c^{k_2} - s)$	Hornsby and Davidson (1973) van Genuchten, *et al.* (1974)
2.4	$\frac{\partial s}{\partial t} = k_r e^{k_2 s} (k_1 c e^{-2k_2 s} - s)$	Lindstrom, *et al.* (1971)
2.5	$\frac{\partial s}{\partial t} = k_r (s_m - s) \sinh \left[k_2 (\frac{s_m - s}{s_m - s_1}) \right]$	Fava and Eyring (1956) Leenheer and Ahlrichs (1971)
2.6	$\frac{\partial s}{\partial t} = k_r c^{k_1} s^{k_2}$	Enfield, *et al.* (1976)

---

* k_1 and k_2 are constants, k_r represents a rate constant (T^{-1}), and s_1 and s_m represent initial and final (or maximum) adsorbed concentrations, respectively (after van Genuchten and Cleary, 1978).

Other relations related to the geometry and boundary conditions of the partial differential equations are also required. These auxillary conditions may include such information as:

- Initial constituent concentrations distributions;
- Type and concentration of potential contaminants;
- Geometry of the waste disposal site;
- Aquifer configuration (2 or 3 dimensional);
- Precipitation/evaporation data; and
- Location of surface water bodies.

Once these relationships have been determined, solutions for constituent concentrations can be obtained by appropriate mathematical manipulations.

MATHEMATICAL SOLUTIONS FOR CONCEPTUAL MODELS

Mathematical solutions for conceptual models are generally obtained by either analytical or numerical approaches, which are briefly discussed in the following text.

Analytical Methods

Most analytical solutions require an assumed constant value for the fluid velocity, dispersion coefficient, and the physical parameters. The advantages of analytical solutions are the ease of application and low-cost of operation, once derived. However, the necessity of having to make simplifying assumptions severely restricts the application of analytical solutions to waste disposal ground-water contamination problems. Although analytical methods have limitations, some of the available two- and three-dimensional analytical solutions (Kuo, 1976; Yeh and Tsai, 1976; Larson and Reeves, 1976) probably could be applied to a few well-defined geohydrological systems (van Genuchten, 1978).

Numerical Methods

A numerical approach is required in most situations because most field problems have such complex physical and chemical characteristics. In most situations, numerical methods may be applied advantageously. Numerical approaches exhibit the advantages of reducing the partial differential equations to a set of approximating algebraic equations. This approach provides a greater flexibility in portraying the complex physical and chemical characteristics of the natural system. Of the numerical methods employed, finite differences, finite element, the method of characteristics, and miscellaneous models are the most common.

Finite Difference--

The finite difference numerical method approximates the derivatives contained in the governing partial differential equations with appropriate difference equations. This approach is normally, successfully employed in ground-water flow studies, but has limited use in ground-water quality studies due to several factors such as:

- Possible introduction of numerical dispersion;
- Occurrence of undesirable oscillations in calculated concentration distributions; especially when the dispersive transport is small when compared to convective transport; and
- Inability to accurately reproduce irregular boundary conditions.

While these inherent limitations exist, finite difference techniques represent the simplest numerical approach especially for simulation of one-dimensional unsaturated transport problems. Two-dimensional solutions are limited. Examples of one- and two-dimensional applications are shown in Table 50.

Finite Element--

The finite element numerical method designates the pressure head and concentration as dependent variables. These dependent variables are approximated and inserted into the governing equation. Resulting errors are minimized through the use of weighted-residual theorems (e.g., the Galerkin Model). As in the case of the finite difference approach, the study area is subdivided into elements which, unlike for finite differences, may assume any configuration required. Finite element methods have been used successfully in a number of field problems involving mass transport, although, in some situations the occurrence of numerical oscillations remain a problem. Examples of linear, triangular, and mixed/higher order finite element applications are shown in Table 50.

Method of Characteristics--

This numerical approach uses a finite difference approach for the flow equation while characteristic equations are used for the constituent transport equation. Characteristics equations are developed by eliminating corrective transport terms from the governing equation and including them in separate equations. The corrective transport terms are reflected by the insertion of "marker" particles (or moving points) into each finite cell. The marker particles are moved through the network as prescribed by the local fluid velocities, thereby establishing the effects of the convective transport terms. The method has been shown to produce acceptable results for a wide variety of field problems for one-dimensional studies (Table 50). An important disadvantage of this method is that it is generally difficult to program in two or three dimensions.

Miscellaneous Numerical Models--

Numerous other numerical models exist which can be applied to ground-water contamination problems. Many numerical models exist which do not rely upon the direct solution of governing partial differential equations. Many of these models do not consider spatial variability in the dependent variables. In this approach, a mass balance equation is generally applied to the entire system, the different input and output variables being only a function of time. (Hornsby, 1973; Gelhar and Wilson, 1974; Donegan and Crawford, 1976; Mercado, 1976).

An approach incorporating more of the characteristics of a distributive parameter scheme may be obtained by directly applying the mass balance equations to a number of well defined spatial cells. By this method, for an explicit time, a finite difference approximation of the governing equations is obtained, provided all significant transport mechanisms are included in the mass balance equations. Tanji, *et al*. (1970) and Orlob and Woods (1967) have presented examples of this method. Elzy *et al*. (1974) applied a similar approach for the development of a vertical-horizontal routing model to the transport of hazardous wastes from a landfill site. A two-dimensional polygonal network has been described by Hassan (1974) in which a multi-layered ground-water basin was modeled.

Each of the numerical techniques discussed above has specific advantages and disadvantages associated with factors affecting the accuracy, efficiency, and accessibility of the particular method. It has been suggested by van Genuchten (1978) that the accuracy of the numerical technique may become secondary when viewed against the uncertainty in the many input parameters needed in most models. Additionally, the programming efficiency, the general setup of the model and its accessibility, are also important factors affecting the usefulness of a particular solution scheme.

SUMMARY

The attenuation and migration of contaminants in the soil/water system are influenced by physical, chemical, and biological mechanisms. The most important physical properties of a soil system relative to leachate migration are diffusion and dispersion, dilution, physical sorption, and straining. Important chemical mechanisms include: precipitation/dissolution, adsorption/desorption, complexation, ion exchange, and oxidation/reduction (redox). Adsorption, complexation, and ion exchange processes are difficult to distinguish from one another;

therefore, they are usually grouped together as if they were one mechanism. Major effects attributed to microbial activities are: oxidation/reduction, mineralization, immobilization, and complexation.

The migration of trace metals in the soil/water system is extremely complex. Thermodynamic and kinetic influences make transport phenomena difficult to explain. Some of the major mechanisms that influence mobility of trace metals are: solubilization, sorption, complexation, and dilution.

The fate of chlorinated hydrocarbons in the environment is very difficult to predict. In general, volatilization, microbial degradation, chemical hydrolysis, oxidation, and sorption can be involved. Various studies indicate that sorption may be the dominant mechanism in soil/water systems.

Factors influencing the fate of pesticides in soil/water systems include: adsorption/desorption, microbial decomposition, votalilization, soil moisture, and physical properties of the soil. Adsorption/desorption is considered to be the prime factor governing the interactions between pesticides and soil colloids.

Recent studies have pointed out that immobilization of viruses by soil should not be equated with virus inactivation because many adsorbed virus particles have been demonstrated to be infectious for significant periods of time. Virus survival within a soil relate to soil moisture content, temperature, pH, nutrient availability, and antagonisms. An evaluation of these factors indicates that viruses which penetrate the soil surface are expected to survive for prolong periods of time, as compared to those retained near the soil surface.

Quantitative data relating to the attenuation of individual constituents by specific mechanisms are not available, consequently, considerable effort has been placed on the development of sophisticated leachate migration models. However, due to the highly complex leachate/soil environments, no leachate migration model exists that can simulate all of the physical, chemical, and biological processes occurring in a typical landfill system. Conceptual-mathematical models appear the most promising, but, unfortunately, also the most complex ones for predicting the performance of waste disposal sites.

SECTION 6

AVAILABLE CONTROL TECHNOLOGY

INTRODUCTION

Environmental control technologies are presented in terms of the following four topic areas:

- Leachate volume control;
- Leachate composition control;
- Leachate collection systems;
- Leachate treatment methods.

Leachate volume control, within the context of this report, refers to the control of water entering the fill or leachates emanating from the fill. Major water sources include contributions from ground water or surface water infiltration.

The largest contribution to landfill moisture, in most situations, is from precipitation. Precipitation can enter the landfill: (1) during the actual filling process in the area of the exposed open working face, (2) as a result of infiltration and percolation of surface runoff and overland flow, or (3) as a result of infiltration from inadequately designed or maintained surface runoff channels either around or through the landfill. Ground water can enter due to a fluctuating water table, or simply due to a poorly selected and designed landfill which has experienced lateral ground-water migration through the fill area. Capillary action through fine grain sands and silts for landfills in close proximity to the ground-water table is another pathway. The landfill operation may also contribute relatively small amounts of water during periods of dust control watering. Available leachate volume control methods for each of these sources are evaluated in terms of the three following evaluation criteria:

- Optimum site conditions most conducive for a particular control approach;
- Limitations associated with the control method; and

- System economics.

Detailed descriptions of control measures, in terms of these criteria, are discussed in the following text as well as presented in tabular form.

LEACHATE VOLUME CONTROL

Ground-water Control Measures

Ground-water control measures are presented in terms of: (1) ground-water/leachate isolation, and (2) leachate plume control.

Ground-water leachate/isolation methods can be defined as those approaches by which contaminants are contained within the fill and not permitted to migrate into the ground water. Leachate plume control refers to those methods which attempt to prevent the contact of the plume of a contaminated ground-water body with additional ground water or surface water resources.

Available control techniques include the following:

- Ground-water/Leachate Isolation
  - Artificial Liners
  - Bottom Sealers
  - Slurry Trench
  - Grout Curtain
  - Sheet Piling Cutoff Wall
- Leachate Plume Control
  - Drains
  - Well Point Systems
  - Deep Well Systems.

Ground-water/Leachate Isolation--

Artificial liners--Artificial or synthetic liners may be utilized as both a collection device and as a means for isolating leachate within the fill and keeping ground water out of the fill. Numerous types of synthetic liners are available and a significant volume of literature exists describing their characteristics (Ewald, 1973; Lee, 1974; Geswein, 1975; Sanks, et al., 1975; Haxo, 1976a,b,c; Haxo, 1977a,b; Haxo, 1978; Stewart, 1978;

Ware and Jackson, 1978; Haxo, et al., 1979; Styron and Fry, 1979; Brown and Anderson, 1980; Matrecon, Inc., 1980). An excellent summary of commonly used synthetic liners and their properties compiled by Kumar and Jedlicka (1973) is presented in Table 53. Structural and physical information for various synthetic liners compiled by Ewald (1973) are provided in Table 54.

To be effective in controlling leachate, liners must possess a number of physical properties. Kumar and Jedlicka (1973) have listed 11 criteria for synthetic liner selection which are listed in Table 55. While abundant literature exists describing these physical characteristics, many properties (e.g., liner reaction to leachates, anticipated liner lifetime for a number of environmental variables, etc.) have not been rigorously quantified. Haxo (1980) provides an excellent discussion of the results of his studies regarding some of these properties.

The installation of a liner for containment purposes has been thoroughly described by Schultz and Miklas (1980). These researchers have categorized installation procedures according to: (1) subgrade preparation, (2) liner placement, and (3) special design considerations.

Subgrade preparation involves compaction and soil sterilization. Compaction of the subsoil may be accomplished with a sheep's foot roller attached to a bulldozer which is suitable for steep slopes. A typical subgrade could consist of 24 in. to 48 in. of compacted soil with a 2-in. layer of sand above the soil. A vibratory roller or compactor may be used to grade these layers. Other approaches have included the use of bulldozers or other earth moving devices for soil compaction, then smoothing prior to liner placement. In those cases where rocks may cause problems due to their size or jagged edges, they are usually removed manually.

Soil sterilization is necessary for controlling the growth of plants which may puncture the liner. The literature indicates that it is imperative that the proper effective regional herbicide be selected for a particular geographic area. It is recommended to wait several days before placing the liner directly over the treated area. Placement of the liner immediately after application of the herbicide is believed to accelerate plant growth process due to increased moisture and warmth (Schultz and Miklas, 1980).

Liner installation requires a number of considerations for successful placement. Among these are site storage, securing liners, bentonite application, seaming, sealing penetrations, wind damage, soil cover on the liners, and quality assurance programs.

TABLE 5.1 PROPERTIES OF COMMONLY USED LINERS (Kumar and Jedlicka, 1973)

Property	Polyethylene		Polyvinyl Chloride	Chlorinated Polyethylene	Poly-propylene	Nylon	Butyl Rubber	Natural Rubber	Hypalon
	Low Density	High Density							
Specific gravity	0.92-0.94	0.94-0.96	1.20-1.5	1.35-1.39	0.9-0.91	1.08-1.4	0.92-1.25	0.91-1.25	
Tensile strength, psi	1.300-2,500	2,400-4,800	3,500-10,000	1800 min.	4000-32000	9,000-11,000	1,000-4000	1,000-3,500	1,000-2,000
Elongation, %	200-800	10-650	60-200	375-575	40-400	250-550			
Shore "A" hardness							15-90	20-100	55-95
Operating temperature range, °F	-70- to 180	-70 to 240	-60 to 200	-40 to 200	-60 to 220	-20 to 380	-50 to 325	-70 to 250	-45 to 250
Resistance to acids	P-G	G	G-E	G-E	G-E	P			G
Resistance to bases	G-E	G-E	G-E	G-E	G-E	E			G-E
Resistance to oxygenated solvents	P-G	P-G	G	P					G
Resistance to aromatic and halogenated solvents	F	F	G	P	G	G	P	P	F
Resistance to aliphatic (petroleum) solvents	P-F	F-G	G	G	G	E	P	P	G
Water vapor permeability, per mils	3-14	1.8-2.2	3-18	0.040-0.048	0.25-1	0.09-1.0	0.15		2.0
Weatherability	P	P	G	E	P	F	G	F	E
Time to crack, hr.	900	300	No crack til 2,500 hr.	No effect til 4,000 hr.	100	1,200			
Time to chalk, hr.	No effect til 2,500 hr.	600	300	Ditto	600	No effect til 2,500 hr.			
Time to fade, hr.	300	300	100	Ditto	900	200			

Note: Data not shown were unavailable to the authors. P = Poor, F = Fair, G = Good, E = Excellent.
ASTM test methods for various properties: specific gravity, D751; Tensile Strength, D97-61T; elongation, D412-61T; Shore "A" hardness, D676-59T; water vapor permeability, E96-66.

TABLE 5.4 TYPICAL CHARACTERISTICS OF SYNTHETIC LINERS (Ewald, 1973)

Base Reinforcement Fabric	Liners of Hypalon Membrane					Liners of EPDM Membrane			Liners of PVC Membrane		
Fabric count	16 x 8	16 x 8	22 x 11	14 x 14	12 x 12	--	--	--	22 x 11	38 x 38	21 x 21
Warp denier	100	100	210	440	840	--	--	--	210	210	840
Filing denier	210	210	420	440	840	--	--	--	420	210	840
Fiber	nylon	nylon	nylon	polyester	nylon	nylon	nylon	nylon	nylon	nylon	nylon
Weave	2/1 leno	2/1 leno	plain	plain	plain	--	--	--	plain	plain	plain
Weight, oz./sq. yd.*	0.4	0.4	1.4	1.8	2.7	2	2	2	1.4	2.0	4.9
Membrane											
Coating thickness-Side A, mils	15	15†	14	14	14	--	--	--	5	4	6-½
Coating thickness-Side B, mils	15	15†	14	14	14	--	--	--	3	2	6-½
Total thickness, mils	30	45	28	28	28	30-34	44-46	61-65	10	10	18
Total weight, oz./sq. yd.	33	48	28-32†	30-32†	30†	--	--	--	10	9	18
Tensile strength, lbs./in.§											
warp	75	120	120-165†	130†	200†	150	150	150	110	200	450
filling	75	120	120-165†	130†	280†	150	150	150	110	190	400
Elongation, % min.											
warp	23	23	--	--	--	30	30	30	--	--	--
filling	23	23	--	--	--	30	30	30	--	--	--
Tear strength, lbs. (tongue)											
warp	20	24	25-50†	10†	70†	25	25	25	40	11	96
filling	20	24	25-50†	10†	70†	25	25	25	40	11	80
Mullen hydrostatic	175	175	225-500†	200†	275†	200	200	200	500+	265	500
	Liners of Butyl Rubber Membrane										
Fabric count	18 x 17	18 x 17	22 x 14	22 x 14	22 x 14	22 x 14	22 x 14	12 x 12	12 x 12	21 x 21	22 x 21
Warp denier	210	210	210	210	210	210	210	840	840	840	840
Filling denier	210	210	420	420	420	420	420	840	840	840	840
Fiber	nylon	nylon	nylon	nylon	nylon	nylon	nylon	nylon	nylon	nylon	nylon
Weave	plain	plain	plain	plain	plain	plain	plain	plain	plain	plain	plain
Weight, oz./sq. yd.	1.0	1.0	1.4	1.4	1.4	1.4	1.4	2.7	2.7	4.8	4.9
Membrane											
Coating thickness, Side A, mils	--	--	--	--	--	--	--	--	--	5-½	--
Coating thickness, Side B, mils	--	--	--	--	--	--	--	--	--	5-½	--
Total thickness mils	17	20	20	--	32	45	62	32	62	16	45
Total weight, oz./sq./yd.	14.5	17.0	18.0	19.4	30	39.0	64	--	--	16	--
Tensile strength lbs./in.											
warp	85	85	100	112	100	100	100	225	--	450	225
filling	70	70	100	107	100	100	100	200	--	375	300
Elongation, % max.											
warp	30	35	20	20	30	30	30	30	--	--	30
filling	30	35	20	18	30	30	30	30	--	--	30
Tear strength, lbs. (tongue)											
warp	5	5	8	12	12	12	12	50	55	40	50
filling	5	5	8	12	12	10-12	12	30	25	35	50
Mullen hydrostatic	100	125	125	185	180	185-190	200	350	--	500+	--

(Continued)

TABLE 5 4 (Continued)

Base Reinforcement Fabric	Liners of Neoprene Membrane							Liners of CPE Membrane		
Fabric count	22 x 14	22 x 14	22 x 14	22 x 14	22 x 14	12 x 12	12 x 12	18 x 17	15 x 15	12 x 12
Warp denier	210	210	210	210	210	840	840	210	440	840
Filling denier	420	420	420	420	420	840	840	210	440	840
Fiber	nylon	nylon	nylon	nylon	nylon	nylon	nylon	nylon	polyester	nylon
Weave	plain	plain	plain	plain	plain	plain	plain	plain	plain	plain
Weight, oz./sq. yd.	1.4	1.4	1.4	1.4	1.4	2.7	2.7	1.0	1.8	2.7
Membrane										
Coating thickness—Side A, mils	—	—	—	—	—	—	—	4	14	14
Coating thickness—Side B, mils	—	—	—	—	—	—	—	4	14	14
Total thickness mils	20	32	32	35	45	32	45	8	28	28
Total weight, oz./sq.yd	20	—	38	52	—	—	—	8.9	30-32	32-34
Tensile strength, lbs./in.										
warp	100	100	100	100	100	225	225	258	130 †	365-420
filling	100	100	100	100	100	200	200	202	130 †	365-420
Elongation, % max										
warp	30	30	30	30	30	30	30	—	—	—
filling	30	30	30	30	30	30	30	—	—	—
Tear strength (lbs./in.)										
warp	10	12	10	10	12	50	50	22	10	58-80
filling	8	10	8	8	10	30	30	22	10	58-80
Mullen hydrostatic	185	180	185	185	190	350	350	185	200	450-475

* 1 oz./yd^2 = 35g/m^2

† This is a 5-ply material, film/scrim/film/scrim/film, with 3 plies of 15-mil-thick film and 2 plies of scrim

‡ 1 lb./in. = 0.18 kg/cm

Notes:
"warp" is the same as machine direction; "filling" is the same as cross direction.
In addition to the above butyl rubber linings, similar linings of EPDM and butyl-rubber/EPDM blends are available.
Weights are without dip.
Unless noted, test methods of ASTM D-751 were used for thickness, total weight, tensile strength, elongation, tear strength, and Mullen hydrostatic data.
The list of PVC liners is far from complete enough to be representative. PVC liners should not be used unless covered with earth or water to prevent rapid weathering degradation.
CPE = chlorinated polyethylene; EPDM = ethylene-propylenediene monomer; and PVC = polyvinyl chloride.

TABLE 55. SELECTION CRITERIA FOR SYNTHETIC LINERS (Kumar and Jedlicka, 1973)

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1. High tensile strength, flexibility, elongation without failure.
2. Resists abrasion, puncture, chemical degradation by leachate.
3. Good weatherability, manufacturer guarantees long life.
4. Immune to bacterial and fungal attack.
5. Specific gravity (S) > 1.0.
6. Color: black (to resist UV light).
7. Minimum thickness, 20 mils.
8. Membrane should have uniform composition, free of physical defects.
9. Withstand temperature variation and ambient conditons.
10. Easily installed.
11. Economical.

Proper storage of the liner is important to minimize deterioration of the liner from the elements. Storage considerations should be predicated upon the type of liner being stored, anticipated weather, length of time the material will be stored at the site, and ambient temperatures. Liner material may be covered with a light or reflective plastic to prevent possible degradation due to sunlight and massive heat buildup.

Securing liners involves the anchoring of the liner to either the subgrade or to penetrations or other concrete structures found within the confines of a surface impoundment or landfill. This may be accomplished by attaching liners in an anchor trench utilizing concrete. Another similar approach utilizes either a slit-trench or a blade-cut type which has a steep edge adjacent to the liner slope and a gently sloping edge away from the liner slope. Fill material is then placed over the liner within the trench. Liner manufacturers usually provide useful approaches for securing liners (Asphalt Institute, 1966; Asphalt Institute, 1974; Portland Cement Assoc., 1974; Staff Industries, Inc., 1976; B. F. Goodrich, Co., 1978, American

Colloid Co., 1979; Burke Rubber Co., 1980; Watersaver Co., 1980).

Bentonite is often used for subgrade cover in areas where compatibility or soil tests show that the proper application will lower the permeability to the desired level. Common application systems include discing, soil spreader, and a light spring toothed rake.

Seaming of the liner materials is an extremely important step. Problems associated with seaming arise when joining the material on steep slopes; moveable support boards or ladders may be used to aid field crew personnel. The use of improper adhesive systems and adverse field conditions (e.g., low temperatures (< 50°F), rain, high winds (> 20 mph) etc.) are other common problems. Selection of the proper adhesive system for a given climate and the identification of a possible back-up adhesive can minimize some of these difficulties. Seaming activities should be halted when the weather conditions are deemed too adverse.

Accumulation of excess liner materials in the landfill corner is another common problem; excess material should be cut and removed. Seaming in the corners and along the slope should always be perpendicular to the toe of the fill (Shultz and Miklas, 1980).

Special precautions must be observed when sealing an intentional penetration (e.g., monitoring wells, grounding cables, etc.) of a liner. A specially manufactured boot may be available which can be sealed to the liner.

Wind damage usually occurs when winds exceed 20 miles/hour. While preventative measures are largely ineffective for winds in excess of 30 miles/hour, damage may be minimized through the employment of various methods. These include anchoring any loose edges with adequately weighted materials or the use of battens (Burke Rubber Co., 1980).

Use of PVC liners require the placement of a soil cover over the liner to minimize the deleterious effects of the sun's ultraviolet radiation. Other liners which are not as succeptible to damage by ultraviolet radiation (Hypalon, HPE, and others) utilize soil covers as a means of stabilizing the liner rather than as a protective measure. Soil covering may also provide a leachate drainage system. For example, the following added sequence of subgrade, liner, and cover soils could serve as drainage medium: (1) smooth graded native material or 0.6 m to 1.2 m (2 ft to 4 ft) of compacted soil; (2) 5 cm (2 in) sand; (3) liner, (4) 5 cm (2 in.) sand; (5) 30 cm (12 in.) of 1-cm (3/8-in.) diameter crushed rock; (6) 15 cm (6 in.) of 1.3-cm to 2.2-cm (1/2-in. to 7/8-in.) diameter crushed rock or gravel; and (7) 15 cm (6 in.) of rock dirt or fine sand (Schultz and

Miklas, 1980; Matrecon, Inc., 1980).

The use of a low permeable barrier material generally requires that a pipe collection network be installed over the liner to collect leachate to prevent ponding and surface runoff. Another function of overdrains is to relieve excessive head buildup and pressure which could result in rupture of the liner. An underdrain system may also be required to collect ground water, preventing pressure buildup and possible rupture of the liner in the event of a rising water table. Collection piping placed beneath liners can also facilitate detection and collection of leachate leakage through a ruptured liner, thus minimizing downward percolation of leachate which could pollute the ground water.

A quality assurance program is an important aspect of any liner installation program. Such a program will include inspection of integrity of seams, proper liner storage methods, wind damage, repair procedures and other activities. Considerations of methods to protect liner edges are especially important if the facility is to be expanded.

In those cases where materials such as asphalt, clay, or bentonite are selected for liners, instatllation procedures will deviate from the installation of a synthetic sheet membrane. Three recommended construction layers for asphalt, clay, and bentonite are listed in Table 56. Specific installation procedures are available from the manufacturers. (Asphalt Institute, 1966; Asphalt Institute, 1976; American Colloid Co., 1979). The applicability and effectiveness of asphalt sealers are discussed in detail in the following section.

Tables 57 and 58 list the updated 1981 installation costs of various liner materials (based on 1977 costs given by Matrecon, Inc. (1980). Of the polymeric membranes--polyethylene (PE) and polyvinyl chloride (PVC) are the least expensive liner materials. Of the soil, admix materials, and asphalt membrane liners--clays, soil + bentonite, soil cements, soil asphalts, and spray-on asphalts are the least expensive to install.

A cost range for ground-water control measures is presented in Table 59 which also summarizes optimum site conditions, and limitations for the employment of artifical barriers.

Monitoring of liner performance is difficult although a number of approaches have been commonly employed. These have included piping laid beneath the liner and underdrains consisting of clay, asphalt, and plastic. These collection systems facilitate detection and collection of leachate passing through a ruptured liner. The collection system is generally drained to a sump or a series of sumps from which the leachate is withdrawn.

TABLE 56. SYNTHETIC LAYERS FOR ASPHALT, CLAY, AND BENTONITE (Asphalt Institute, 1966; Middlebrooks, et al., 1978)

	Asphalt	Clay	Bentonite
Subgrade	24-48"* of graded, compacted soil.	24-48" compacted soil.	12" of clean soil.
Liner	2" of hot rolled asphalt.	24" of compacted, impervious material placed in one 12" layer and two 6" layers.	Mix bentonite with clean soil in desired ratio. Disc to depth of 2-3" (no sheeps foot allowed).
Cover	12" of 3/4" diameter crushed rock. 6" of 1/4-3/4" diameter crushed rock. 6" of rock dust or fine sand.	12" of 3/4" diameter crushed rock. 6" of 1/4" or 3/8" diameter crushed rock. 6" of rockdust or fine sand.	12" of compacted imported earth.

* 1 in. = 2.5 cm

TABLE 57. COSTS OF FLEXIBLE POLYMERIC MEMBRANE, PLASTIC, AND RUBBER LINERS, 1981 (after Matrecon, Inc. 1980)

Material	Nominal Thickness mils	1981 Installed Costs $/yd^2*
Butyl rubber	30	4.90 - 5.20
Chlorinated polyethylene (CPE)	30	4.50 - 5.00
Chlorosulfonated polyethylene (CSPE)	30	4.50 - 5.00
Elasticized polyolefin (ELPO)	20	3.00 - 3.25
Ethylene propylene rubber		
(EPDM) covered	30	4.90 - 5.00
uncovered	30	4.15
Neoprene (CR)	30	8.40 - 9.70
Polyethylene (PE)	10	1.60 - 2.00
Polyvinyl chloride (PVC)	20	2.10 - 2.85
	30	2.60 - 3.50

* $1.00/yd^2 = $1.20/m^2

Because of the possibility of liner leakage, placement of a small piece of the installed liner (coupon) in the fill may be used to evaluate the effects of the waste on the liner material properties over time intervals. Coupon placement should allow for essentially the same exposure to the waste and environmental conditions as the installed liner.

Bottom sealers--A bottom seal is a method by which a sealing material is applied on the bottom of a new landfill site or a pressure-injected grout is pumped under an existing site. A variety of materials, such as asphalt concrete, soil asphalt, soil cement, sprayed asphalt membranes, and bituminous seals, may be utilized for sealing purposes, depending upon soil properties (Haxo, 1977a; Tolman, et al., 1978; Middlebrooks, et al., 1978; Matrecon, Inc., 1980). This approach requires a detailed knowledge of both landfill boundaries and soil properties; extensive exploratory corings and soil testings are therefore required with this method. Literature describing bottom sealing is extensive due to its use in construction (Karol, 1968; Lenahan, 1973; Halliburton Services, 1976; American Cyanamid Co., 1975).

TABLE 58. COST ESTIMATES OF SOIL, ADMIX MATERIALS AND ASPHALT MEMBRANE LINERS, 1981 (after Matrecon, Inc. 1980)

Liner Type	1981 Installed Costs $/yd^2*
Compacted clay, 2-ft thick	2.30
Soil + bentonite, 6-in. thick 9 lb/sq yd	1.45
Soil cement, 6-in. thick + sealer (2 coats - each 0.25 gal/sq yd)	2.50
Soil asphalt, 6-in. thick + sealer (2 coats - each 0.25 gal/sq yd)	2.50
Asphalt concrete, dense - graded paving with sealer coat (hot mix, 4 - in. thick)	4.65 - 6.45
Asphalt concrete, hydraulic (hot mix, 4-in. thick)	6.00 - 8.30
Bituminous seal (catalytically blown asphalt) 1 gal/sq yd	3.00 - 4.00 (with earth cover)
Asphalt emulsion on mat (polypropylene mat sprayed with asphalt emulsion)	2.50 - 3.50

* $1.00/yd^2 = $1.20/m^2

Installation costs for bottom sealers vary according to soil porosity. For pressure-injected sealing, a 1.2-m (4-ft) thick portland cement sealer for a 4-hectacre (10-acre) landfill with a porosity of 20% may range in cost from $1,610,000 to $4,010,000; at 30% porosity, from $3,470,000 to $6,020,000 in 1981 dollars. For the same area, a 1.8-m (6-ft) thick liner (20% porosity) may range in cost from $2,400,000 to $6,000,000; at 30% porosity, $3,600,000 to $9,000,000 in 1981 dollars (after Tolman et al., 1978).

Bottom sealing practices offer advantages over other alternatives (primarily excavation and re-burial) especially in coarse grained soils and gravels. The seal can prevent ground water

TABLE 5 9 .COSTS OF GROUND-WATER CONTROL MEASURES, 1981 (after Tolman, et al., 1978; Matrecon, Inc., 1980)

Control Technique	Optimum Site Conditions	Limitations	1981 Installation Costs	Perceived Effectiveness
Artificial Liner	Ground water near to the refuse/soil interface. Underlying soil possess a moderately rapid to very rapid permeability 1.4×10^{-3} cm/sec to $> 1.4 \times 10^{-2}$ cm/sec.	Requires installation prior to landfilling. Expected life not established. Rising ground water may rupture the liner. May be damaged during landfilling activities. Once installed, extremely difficult to repair.	\$1.60 to \$9.70 per yd^{2}[*]	Effective when used as a leachate collection method or as a means for isolating leachate within the fill and keeping ground water out of the fill.
Bottom Sealer	Same as for artificial liner and when a detailed understanding of the soil is known.	Detailed understanding of the soil properties are required. Methods of determining that all voids have been effectively grouted are not available. Leachate may have a deleterious effect on grout integrity.	\$1.45 to \$8.30 per yd^{2} (See Table for design thickness)	Same as for artificial liners.
Slurry Trench	Near to surface ground water and bedrock condition. Other conditions similar to those for an artificial liner.	Costs of shipping bentonite. Excavation difficult due to soil conditions. Deterioration of bentonite when exposed to high strength leachate may occur.	\$420 to \$710 per lineal ft[†][‡]	Effective underground-water barrier used to divert ground-water flow around a landfill site or impede the horizontal subsurface movement of leachate from the fill.
Grout Curtain	Same as for artificial liner as well as a near to surface bedrock condition.	Costs may be prohibitive for large scale use. Difficult to assess the integrity of the seal.	\$680 to \$1690 per lineal ft.	Performs similar function as slurry trench. Considered effective in soils with permeabilities of 10^{-5} cm/sec or greater.
Sheet Piling Cutoff Wall	Same as for grout curtain.	Corrosion potential and problems with driving the sheet through rocky soils, ability to maintain integrity of the piling.	\$550 to \$820 per lineal ft.	Effective control measure when a deep vertical barrier is required to retard the flow of ground water under and through a landfill (no excavation is required).

[*] \$1.00/yd^{2} = \$1.30/m^{2}

[†] 1 ft = 0.31 m

[‡] Lineal ft costs based on a 518 m (1700 ft) long structure, 18 m (60 ft) deep.

from entering the landfill as well as impounding leachates formed within the fill. Disadvantages cited for bottom sealing include problems associated with exploratory drilling, over-design to compensate for the erratic nature of filling, and the uncertainty of determining whether a complete seal has been formed (Tolman, et al., 1978).

A wide variety of admix and asphalt sealing materials may be used as bottom sealers. Primary advantages of asphalt materials include universal availability, versatility in available forms, low cost, and the fact that asphalt materials are some of the most logical engineering materials available for large scale installations. Utilizing pure asphalt in membrane form has proven effective in seepage control and cost effectiveness. The following are examples of commonly used bottom sealers.

Asphalt concrete is a mixture of asphalt cement and graded aggregate that is placed and compacted at elevated temperatures. This sealer is well developed for use in construction of lining for all types of hydraulic structures. When properly mixed and applied, asphalt concrete forms a durable, stable, and erosion-resistant lining. Mix design of asphalt cement should follow guidelines set by the Asphalt Institute (1974). The expense for this type of membrane is much higher than natural clay liners (Table 58),but is comparable with synthetic types (Table 57).

Soil asphalt embrace a wide variety of soils, usually those of low plasticity, mixed with a liquid asphalt. Soil asphalts containing emulsified asphalts usually require a waterproofing seal, membrane, or asphalt cement, placed on top of them. Other types of asphalt have been used with soils to provide an effective soil asphalt which is more flexible and crack resistant than soil cement or asphalt cement, and more resistant to aging, as well (Matrecon, Inc., 1980). Soil asphalt is a relatively inexpensive bottom sealer (Table 58).

Soil cement is prepared by mixing portland cement, water, and a variety of soils. As the cement hydrates, the mixture hardens resulting in an impermeable liner. The permeability of the sealer is dependent upon the nature of soil used in the mixture. The more granular the soil, the higher the permeability. Usually, surface sealants are applied to soil cement to impart waterproof capabilities. Aging and weathering characteristics are usually good, especially those associated with the wet-dry and freeze-thaw cycles. Some degradation of the cement is expected in acidic environments (Stewart, 1978). Costs are comparable to soil asphalt (see Table 58).

Sprayed asphalt membrane sealer (hot-sprayed type) usually consists of a continuous layer of asphalt, often without filler or reinforcement, covered or buried to provide protection from weathering or mechanical damage. Cover soils may include native soil, gravel, or asphalt macadam. These membranes are then dried into tough, pliable sheets that conform readily to soil contour alterations. Useful lives of 15 years or greater have been observed when these membranes are carefully applied and covered with an adequate layer of fine grained soil (Middlebrooks, et al., 1978). Sprayed asphalt membranes are one of the least expensive types of current bottom sealers (see Table 58).

Bituminous seals are most often used to seal the pores of an asphalt mixture serving as a liner in order to provide waterproofing. There are basically two types of bituminous seals. One is called catalytically-blown asphalt since it is produced by means of air-blowing the asphalt in the presence of a catalyst (usually ferric chloride) which produces a brownish tint. The asphalt produced has a high softening point, yet remains flexible, tough,and impervious to water. It is resistant to acids, but not oils. Aging resistance is also good (Lutton, et al., 1979). The second type of butiminous seal is the fabric plus asphalt emulsion type. These form continuous films of asphalt after breaking of the emulsion and the evaporation of the water. These films are less tough than films of the aforementioned type; therefore, toughness is achieved by spraying the asphalt on supportive fabric such as glass fiber, woven jute, etc. (Geswein, et al., 1978). The combination has been used quite successfully as a lining for ponds. The cost is relatively low for this type of lining (see Table 58).

Slurry-trench--A slurry-trench is an underground water barrier which may be used to prevent the horizontal subsurface movement of leachate from a landfill or to impede the movement of water into the fill (Tolman, et al., 1978). The slurry-trench cutoff wall may isolate leachate from a landfill by providing a complete seal extending to an impermeable liner or by increasing the up-gradient length of the ground-water flow path. The slurry-trench may also be utilized as a down-gradient barrier from the landfill, in which case, leachate impounded by the barrier may be collected for recycling and/or treatment. Slurry-trench technology has been developed for use in the construction industry and considerable literature exists describing this application along with sanitary landfill applications (Seaman, 1966; Gerwick, 1967; Kapp, 1969).

Tolman, et al. (1978) developed estimated costs for slurry-trenching. A completed slurry-trench 18 m (60 ft) deep and 1 m (3 ft) wide will range from $1,380 to $2,330/m ($4.20 to $7.10/ lineal ft) based upon 1981 costs. To construct a semicircular cutoff wall 518 m (1,700 ft) long around the up-gradient end of a 4-hectacre (10-acre) landfill would cost between $720,000 and $1,200,000 (1981 dollars). Differences in price depend on whether the excavation is soil or rock and the type of bentonite slurry used. Emrich, et al. (1980) proposed a slurry-trench cutoff wall 945 m (3100 ft) long and 44 m (145 ft) deep to help protect ground waters from a sanitary landfill in Windham, Connecticut. A cost range of $3.7 to $5.6 million was proposed for the 10-ha (25-acre) site (1981 dollars).

Advantages of slurry-trenches include relatively simple construction methods, availability of leachate-resistant bentonites, low maintenance, and minimal liner deterioration. Problems associated with slurry-trenches include the cost of shipping bentonite, over-excavation in rocky ground due to boulders, and the possible deterioration of certain bentonites when exposed to high ionic strength leachates. Slurry-trenches are most effective in those cases in which a shallow ground-water condition exists and a impermeable soil or bedrock strata is available for contact with the barrier (Tolman, et al., 1978).

Grout curtain--A grout curtain performs a similar function as that of a slurry-trench in providing a vertical barrier to either divert ground-water flow around a landfill site or intercept migrating leachate. A grout curtain is created by injecting solutions under pressure into soils and underlying earth materials. The grout solution fills soil voids preventing the passage of leachates or ground-water flow (Tolman, et al., 1978).

Grout materials include cement, bentonite, epoxy resins, silicone rubbers, lime, fly ash and bituminous compounds (Raymond International, Inc. 1967; Tolman, et al., 1978). The use of a particular grout material is a function of soil characteristics encountered at the facility. American Cyanamid Co. has categorized the use of chemical grouts with various soil properties (American Cyanamid Co., 1975).

The variety of grout materials available in a particular situation will result in a disparity of costs. The relative costs of grouts in comparison to portland cement are listed in Table 60. Tolman, et al., 1978 developed estimated costs for portland cement grout curtains. An installed grout curtain 18 m (60 ft) deep and 1.5 m (5 ft) wide would cost from $2,230 to $5,540/m ($680 to $1,690/lineal ft) based on 1981 dollars. Estimated costs for a 518-m (1700-lineal ft) portland cement grout curtain up-gradient of a 4-hectare (10-acre) landfill to a depth of 18 m (60 ft) would cost between $1,150,000 and $2,880,000 (1981 dollars).

Grouting technology is well developed and is applicable to a wide range of soil properties. However, grouting is usually considered effective only in soils with permeabilities of 10^{-5} cm/sec or greater (Tolman, et al., 1978). Grouting offers a useful approach in consolidated or cohensionless soils although costs may be prohibitive for large landfills. The use of grout curtains for diverting ground water around landfills or intercepting leachates is undocumented; however, the principle should be applicable for landfilling practices.

TABLE 60. RELATIVE COSTS OF GROUT* (Kays, 1977)

Type of Grout	Basic Cost Figures
Portland Cement	1.0
Silicate Base-15%	1.3
Lignin Base	1.65
Silicate Base-30%	2.2
Silicate Base-40%	2.9
Urea Formaldehyde Resin	6.0
Acrylamide (Am-a)	7.0

* Base unit = 1. Under a given set of conditions, where portland cement grouts costs 1.0 times $/unit, other types of grout will cost the given figure times $/unit.

Sheet piling cutoff wall--Sheet piling cutoff walls may be used to retard the flow of water under and through a landfill. A sheet piling cutoff wall consists of lengths of steel sheet piling permanently driven into the ground. Sheet piling is readily available in various shapes and weights, the selection of which is based on costs and ease of installation. The sheet piling is assembled before being driven into the ground. The sheet piling wall is not watertight due to mill tolerances in the interlocking edges. Cathodic protection can prolong the useful life of the wall, which can range up to 40 years (Tolman, et al., 1978).

Tolman, et al. (1978) developed costs for a 518 m (1700 ft) long and 18 m (60 ft) deep cutoff wall up-gradient of a 4-hectacre (10-acre) landfill. Total costs to construct and install a 20 ton sheet piling cutoff wall (including shipping costs from Buffalo, New York to Philadelphia) are estimated to be between $940,000 to $1,380,000 based on 1981 dollars. Installed costs range from $1,800 to $2,690/m ($550 to $820/lineal ft). The unit cost of the sheet metal plus shipping costs are major variables in any cost estimates for installed sheet piling cutoff walls.

Sheet piling cutoff approaches offer several distinct advantages over slurry trenches and grout curtains. Among these are ease of construction (no excavation is necessary), availability of contractors to perform the work, maintenance free nature of the wall, and the relative economical construction costs. Limitations of the sheet piling cutoff wall include its corrosion potential, problems of driving the sheet through rocky ground, and that the sheet piling is not initially watertight (Tolman, et al., 1978).

Leachate Plume Control--

Leachate plume control refers to those efforts to prevent a contaminated ground-water body from contacting additional groundwater or surface water resources. Methods involve the removal of water by pumping or drainage from around the landfill. These approaches include drains, well point systems, and deep well systems.

Plume control is similar in purpose to subsurface infiltration barriers by effecting changes in water table elevations, and are often used in conjunction with subsurface barriers. The control measures differ insofar as a continuing input of energy is usually required for pumping water. Table 61 summarizes these systems. These systems may be utilized either up-gradient or down-gradient from a landfill.

Drains--The purpose of drains is to intercept up-gradient ground water which can be redirected from the landfill to lower the water table. They can also be used to collect ground water and leachate down-gradient of the landfill. Drains are constructed by excavating a trench to a desired depth, partially backfilling the trench with highly permeable sand or gravel, placing a plastic or ceramic drain tile in the sand and gravel bed, and complete the backfilling. The drainage can be collected in an impermeable sump and directed or pumped back to the landfill.

Tolman, et al., 1978 developed estimated capital costs for ground-water drainage by trenches or subgrade drains. Based on 1981 dollars, the costs would range from $1,780 to $2,490 per hectare ($720 to $1,000 per acre) or $11.30 to $15.90/m ($3.40 to $4.85/lineal ft), depending on the depth of placement and materials used. Operating and maintenance costs are estimated to be $2,230 per year or $3.48/m ($1.06/lineal ft).

Drainage systems are especially useful in situations where high ground-water conditions exist, providing a relatively efficient control system in these cases. Disadvantages include difficulties in trenching operations for coarse or rocky soils, as well as problems associated with deep trenching.

Well point systems--Well point systems can be placed up-gradient from a landfill to prevent ground-water flow through the fill. Well points may also be used for leachate collection down-gradient of the fill, although care must be exercised to prevent the system from accelerating the rate of leachate migration from the fill.

TABLE 61. LEACHATE PLUME CONTROL (Tolman, et al., 1978)

CONTROL TECHNIQUE	OPTIMUM SITE CONDITIONS	LIMITATIONS	1981 COSTS	PERCEIVED EFFECTIVENESS
Drains	Near-to surface ground water; soils which are easily excavated, consistent ground water level with little fluctuation.	Ground water may be too low for effective use of drains. Soils may not be amenable to excavation. Perimeter area required for drainage may be too large an area for practical drainage purposes.	Capital costs $3.40 to $4.85 per lineal ft.[*] Operating and maintenance costs $1.06 per lineal ft per yr.	Effective when used in conjunction with other control approaches such as liners, well points, etc.
Well point system	Same as for drains	Usually effective for shallow dewatering purposes (<10m)	Capital costs $13 to $14 per lineal ft. Operating and maintenance costs[†] $24 per lineal ft per yr.	Effective when used to compliment another control system (e.g., leachate treatment).
Deep well system	Water exists at greater depths than a well point system can withdraw.	Capable of withdrawing approximately 40 vertical ft of water in uniform sands.	Capital costs $22 per lineal ft. Operating and maintenance costs[†] $22 per lineal ft per yr.	Lowers the water table in deeper applications than a well point system.

[*] 1 ft = 0.31 m

[†] Power costs included

Well points are short lengths of plastic (PVC) or telfon well screen which can be placed in the saturated zone up- or down-gradient from the fill. The well points are connected to a pump which evacuates the air from well points creating a vacuum which forces the ground water to flow through the well points. Well point systems are usually effective for shallow dewatering purposes up to 10 m (32 ft), and may provide draw-downs of up to 4.5 m (15 ft) within the pumping zone of influence (Nobel, 1963).

Based on costs reported by Tolman, et al. (1978), well point systems are estimated to cost between $43 to $48/m ($13 to $14/lineal drain-ft) for installation, plus $2,880 to $4,320 for a 19 to 24-liter/sec (300-400 gal/min) pump, based on 1981 dollars. To drain a 275-m (900-lineal-ft) area up-gradient from a landfill would cost approximately $17,300 (capital cost). This capital cost, in addition to pumping tests, analyses and engineering costs, could result in a $31,000 to $40,300 total capital expenditure. Maintenance, power, and labor costs for this system are estimated to be approximately $21,600 per year or $79/m ($24/lineal ft) per year. Power costs represent the major expense (i.e., $15,000/yr. based upon a 32 kw (24 hp) demand at $0.10/kwh.

The advantages of a well point system are that installation costs are moderate and the construction methods are relatively simple. The disadvantage of this system is that the power costs are high, continued maintenance is necessary, and a long term financial commitment is required.

Deep well systems--Deep well withdrawl refers to an installation in which a greater vertical depth of water can be removed than can be handled by a vacuum-extracted well point system. Deep wells are capable of lowering the water table as much as 12 m (40 ft) in uniform sand. Each well is usually equipped with its own submersible pump and is, therefore, capable of dewatering large areas.

Deep well dewatering systems generally require a higher initial investment than well point systems, since they require more materials, equipment, and greater installation costs. Based on costs given by Tolman, et al. (1978), it is estimated that deep well systems (1981 dollars) would cost $72/m ($22/lineal ft) for a 10-cm (4-in) PVC well; a screen may cost $140 and a pump $1,140 to $1440. Capital costs for a well system 8 m (60 ft) deep to dewater a 275 m (900 ft) front is estimated to cost approximately $25,900. Maintenance, power, and labor costs for this system are estimated to be approximately $20,200 per year or $73/m ($22/lineal ft) per year. Power costs represent the major expense (i.e., $15,000/yr based upon a 32 kw(24hp) demand at $0.10/kwh).

Emrich, et al. (1980) proposed a deep well dewatering system to lower the water table surrounding a sanitary landfill. Estimated capital costs for a ring of 80 wells, 19.5 m (65 ft) deep, generating 25.1 l/sec (400 gpm) are $198,000 in 1981 dollars; operation and maintenance costs for such a system are estimated at $17,000 to $19,000 per year.

Surface Water Control

Surface water control is an approach by which infiltration of water entering the landfill is reduced or prevented altogether. By utilizing control measures to minimize the quantity of water entering the landfill via rainfall or runon, leachate generation can be reduced. This section presents three methods of reducing infiltration:

- Contour grading and surface water diversion;
- Surface sealing; and
- Revegetation.

Contour Grading and Surface Water Diversion--

Grading and compacting the landfill to a profile of a maximum of 12 percent and a minimum of 6 percent will allow surface water to draim from the site and will minimize infiltration (Tolman, et al., 1978). Most soils will remain stable at these grades; however, slopes greater than 12 percent may accelerate erosion of the landfill, depending on the soil type and extent of vegetative cover (U.S. Dept. of Agriculture, 1975). Ideally, to minimize overland routing areas subject to infiltration, the center of the landfill should be the highest elevation. Exact slopes to be used should be designed on the basis of soil type and slope stability at the site (Tolman, et al., 1978).

Contour grading operations are simple and economical. Costs for contour grading are affected by the size of the landfill, availability of cover material, and the percent of slope. Based on costs developed by Tolman, et al. (1978), the total cost of contour grading a 4-hectacre (10-acre) landfill plus a 0.6 m (2 ft) minimum thickness of soil cover would range from $181,000 to $349,000 (1981 dollars).

Contour grading is a common practice for sanitary landfills. Emrich, et al. (1980) studied seven possible remedial actions to minimize or abate pollution from a closed landfill in Connecticut. It was concluded that contour grading, in conjunction with diversion ditches, a PVC membrane seal, and a vegetative cover, would be the most effective and economical control measures. Monitoring of the landfill will continue for two years following the installation of the remedial action alternatives to determine their effectiveness.

Diversion ditches are usually constructed around the perimeter of a site for diversion of upland rainfall, as well as for re-routing water from spring-heads. Costs for diversion ditches vary according to ditch configuration, placement of material within the ditch, and the use of a liner material (e.g., plastic, concrete culvert, etc.). Diversion ditches are generally designed for a 10-year storm intensity (Tolman, et at., 1978). Figure 46 presents some design specifications for surface and interceptor ditches.

Updated 1981 cost estimates based on Tolman et al. (1978), for a 0.6 m (2 ft) deep and a 1 m (3 ft) wide trench range from \$54.00 to \$86.40 per m (\$16.30 to \$26.10 per lineal ft). Included in this figure is an assumed cost of \$11.40 to \$16.90/m^3 (\$8.60 to \$13.00/yd^3) for trench excavation.

Diversion ditches and surface contouring represent two mitigations which are extremely cost effective control approaches. Table 62 summarizes these methods and other surface water control measures for a 4-hectacre (10-acre) landfill.

Surface Sealing--

Surface sealing is a method of landfill closure by which an impermeable material is used to cap or seal a landfill, preventing water infiltration and minimizing leachate generation. Caps and seals can be constructed of clays, fly ash, soil-cement, lime-stabilized soil, membrane liners, bituminous concrete, and asphalt-tar materials. Membrane and asphalt compounds may require an earth cover to protect the cover seal and to provide support (Tolman, et al., 1978). In addition to the above-mentioned materials, 114 chemical additives which may be mixed with a soil cover for various beneficial effects (e.g., stabilization, repel water or resist water erosion, or resist dusting or wind erosion) are presented (Lutton, et al., 1979).

Lutton, et al. (1979) provides a four step process for selection and design of cover as well as the functions which a cover provides. Specific functions of cover on waste disposal sites include the following:

- Control insect emergence and entrance;
- Control rodent burrowing;
- Reduce bird and animal attraction;

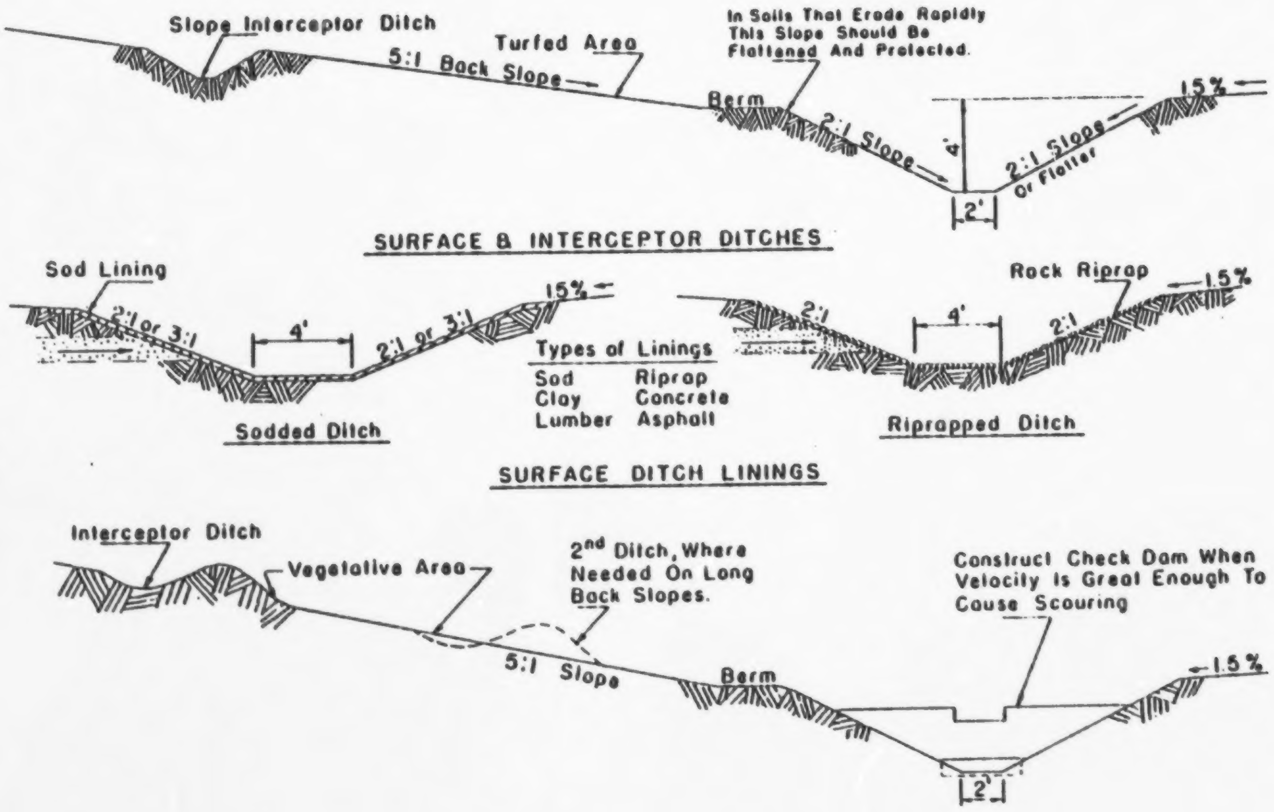

Slope Interceptor Ditch
Turfed Area
5:1 Back Slope
In Soils That Erode Rapidly This Slope Should Be Flattened And Protected.
Berm
2:1 Slope
4'
2:1 Slope Or Flatter
1.5%
2'
SURFACE & INTERCEPTOR DITCHES
Sod Lining
2:1 or 3:1
4'
2:1 or 3:1
15%
Types of Linings
Sod Riprap
Clay Concrete
Lumber Asphalt
Sodded Ditch
Rock Riprap
2:1
4'
2:1
1.5%
Riprapped Ditch
SURFACE DITCH LININGS
Interceptor Ditch
Vegetative Area
2nd Ditch, Where Needed On Long Back Slopes.
5:1 Slope
Berm
Construct Check Dam When Velocity Is Great Enough To Cause Scouring
1.5%
2'

Figure 46. Design specifications for surface and interceptor ditches (Lutton, et al., 1979).

TABLE 6 2 . SURFACE WATER CONTROL (Tolman, *et al*., 1978)

Control Techniques	Optimum Site Conditions	Limitations	1981 Costs[a]	Perceived Effectiveness
Contour Grading and Surface Water Diversion	Landfill stabilized; sufficient borrow area for soil excavation available nearby.	Available soil cover should be present. Differential settlement may be extensive, requiring large volumes of soil.	$181,000 to $349,000	An effective control measure used both during landfilling and once the fill has been stabilized.
Surface Sealing	Borrow material available; landfill has been stabilized and/or graded.	Suitable cover material may not be readily available. Gas venting may also be required.	$202,000 to $829,000	Effective if settlement is minimal and when used in conjunction with other control measures.
Revegetation	Stabilized fill and available cover soil (> 2 ft). Minimum of settlement, and gas control equipment available at the site.	Decomposition gases may be toxic to vegetation if not vented or controlled.	$10,500 to $17,300	Provides a buffer to moisture entering the fill, and reduces erosion. Is most effective when employed in conjunction with other control measures.

[a] Costs are based on a 4-hectare (10-acre) site

- Minimize water erosion;
- Minimize water infiltration;
- Minimize or maximize gas containment;
- Minimize fire hazard potential;
- Minimize blowing paper;
- Control noxious odors;
- Provide sightly appearance;
- Minimize settlement and maximize compaction;
- Provide for vegetative growth;
- Minimize wind erosion and dust generation;
- Resist cold climate deterioration and operational difficulties;
- Preserve slope stability; and
- Resist cracking.

Emrich, et al. (1980) reported the installation of a 20 mil PVC membrane seal for a sanitary landfill which required 10 cm, (4 in.) of fine grained sand as a subgrade. A cover for the membrane consisted of a 46 cm-(6-in.) layer of sand and gravel placed over the membrane. Lutton, et al.(1979) describes the advantages of several cover materials layered over a membrane which is not obtained with a single material. A vertical cross-section of such an approach is as follows: (1) refuse, (2) sand (buffer), (3) silt (filter), (4) clay (barrier), and (5) a loam. The cover profile for a particular site may vary according to the climatic conditions, availability of materials, design considerations, etc.

Costs for sealing depends upon the amount of earth-moving and grading as well as the selection of seal materials. Tolman, et al. (1979) developed costs for landfill covers for a variety of seal material for a 4-ha (10-acre) site. These costs (1981 dollars) are listed in Table 63.

Surface seals offer a number of advantages in surface water control. Among these benefits are: (1) surface seal placement is economical and is relatively simple to install, (2) many of the cover materials can withstand minor settlement, and (3) some covers such as soil-cement and bituminous pavings do not require soil covers and long service life is expected.

TABLE 63. COSTS OF SURFACE SEALS (Lutton, et al, 1979)

Material	Thickness		1981 Cost Range ($)*†
	cm.	in.	
Clay Cap	15	6	201,700 - 368,900
Clay Cap	46	18	259,800 473,300
Bituminous Concrete Cap	4	1.5	277,200 - 490,800
Bituminous Concrete Cap	13	5	416,600 - 630,100
Fly-Ash Cap	30	12	196,000 - 356,100
Fly-Ash Cap	60	24	258,600 - 481,500
Soil-Cement Cap	13	5	301,600 - 462,800
Lime-Stabilized Cap	30	5	301,600 - 462,800
PVC Membrane Cap	30 mil	0.030	559,900 - 828,700

* Cost estimates reflect land preparation costs for cover placement, including:

- excavation of 26,700 m^3 (34,950 yd^3) of common borrow material;
- excavation of 7,700 m^3 (10,000 yd^3) of the waste during grading operations; and
- earthmoving operations, grading and compacting materials, equipment and procedures for constructing each seal (estimated at $1440 per acre).

† Costs are based on a 4-hectacre (10-acre) site.

Problems associated with cover seals include the unavailability of certain cover material, such as natural clay deposits; large quantities of borrow material may be required; and gas venting is required with all surface seals.

Revegetation--

Revegetation provides an inexpensive and cost-effective means of stabilizing the landfill cover, reducing infiltration, and help minimize erosion by wind and water. Revegetation can also reduce water runoff resulting in more water entering the soil. This increased infiltration is offset at least partly by transpiration from vegetation, through there is considerable question with regard to the relative importance of these offsetting processes (Rovers and Faquhar, 1973; Motz, 1974). A subsidiary benefit is that revegetation enhances the appearance of the landfill site. A cover layer of suitable soil at least 0.6 m (2 ft) thick is usually required for rooting purposes (Tolman, et al., 1978).

The selection of the best revegetative cover for a particular site depends upon soil type, nutrient, pH levels , climate, and knowledge of plants that have the desired characteristics. If soils do not possess sufficient nutrients to support vigorous plant growth, sludge might be used as a soil conditioner. Bennett and Donahue (1975) and Lutton, et al.(1979) provide comprehensive descriptions of revegetation practices. Many species of grasses and legumes are used for revegetation. Commonly used species are described in Table 64.

One of the disadvantages of revegetation is that an impervious clay or plastic barrier incorporated in the cover makes the plant-root zone susceptible to swamping after moderate rains since verticle drainage is impeded (Lutton, et al, 1979). Irrigation may be necessary during prolonged dry spells to prevent loss of plant cover. Also, revegetation may be damaged by landfill gases which continue to vent for years after site closure (Flower, et al, 1977). Tolman, et al. (1979) prepared costs for vegetating a completed fill. Based upon a slope of less than 12% and a number of seeding and cover soil preparation assumptions, costs would range from \$0.17 to \$0.26 per m^2 (\$0.15 to \$0.22 per yd^2) in 1981 dollars. Mulching with straw or hay would cost between \$0.09 and \$0.17 per m^2 (\$0.07 and \$0.15 per yd^2). From these cost estimates, a cost range of \$10,500 to \$17,300 for a 4-ha (10-acre) landfill was derived (1981 dollars).

LEACHATE COMPOSITION CONTROL

The major objective of leachate composition control is to reduce the strength and contaminant flux of leachate. Leachate composition is a function of many factors including those inherent in the refuse mass and landfill location, and those

TABLE 6 4 GRASSES AND LEGUMES COMMONLY USED FOR REVEGETATION (Lutton, 1980)

Variety	Best Seeding Time	Important Characteristics	Area/Conditions of Adaptation
Grasses			
Redtop bentgrass	Fall	Strong, rhizomatous roots, perennial	Wet, acid soils, warm season
Smooth bromegrass	Spring	Long-lived perennial	Damp, cool summers, drought resistant
Field bromegrass	Spring	Annual, fibrous roots, winter rapid growth	Cornbelt eastward
Kentucky bluegrass	Fall	Alkaline soils, rapid grower, perennial	North, humid, U.S. south to Tennessee
Tall fescue	Fall	Slow to establish, long-lived perennial, good seeder	Widely adapted, damp soils
Meadow fescue	Fall	Smaller than tall, susceptible to leaf rust	Cool to warm regions, widely adapted
Orchard grass	Spring	More heat tolerant, but less cold resistant than smooth bromegrass or Kentucky bluegrass	Temperate U.S.
Annual ryegrass	Fall	Not winter hardy, poor dry land grass	Moist southern U.S.
Timothy	Fall	Shallow roots, bunch grass	Northern U.S., cool, humid areas
Reed canarygrass	Late Summer	Tall coarse, sod former, perennial, resists flooding and drought	Norhtern U.S., wet, cool areas
Legumes			
Alfalfa (many varieties)	Late Summer	Good on alkaline loam, requires good management	Widely adapted
Birdsfoot trefoil	Spring	Good on infertile soils, tolerant to acid soils	Moist, temperate U.S.
Sweet clover	Spring	Good pioneer on non-acid soils	Widely adapted
Red clover	Early Spring	Not drought resistant, tolerant to acid soils	Cool, moist areas
Alsike clover	Early spring	Similar to red clover	Cool, moist areas
Korean lespedeza	Early spring	Annual, widely adapted	Southern U.S.
Sericea lespedeza	Early Spring	Perennial, tall erect plant, widely adapted	Southern U.S.
Hairy vetch	Fall	Winter annual, survives below 0°F, widely adapted	All of U.S.
White clover	Early Fall	World-wide, many varieties, does well on moist, acid soils	All of U.S.
Crownvetch	Early Fall	Perennial, creeping stems and rhizomes, acid tolerant	Northern U.S.

created by landfill construction and operational features. The chemical and biological properties of refuse are largely uncontrollable; however, the codisposal of sewage sludge or industrial wastes with municipal refuse may increase refuse stabilization. Available leachate composition control techniques include the following:

- Landfill construction and operational features;
- Leachate recirculation;
- Addition of municipal sludge or industrial wastes; and
- Addition of selected sorbents.

Landfill Construction and Operational Features

Landfill and operational features which are significant in influencing leachate composition are the physical characteristics of the refuse, including particle size (shredding) and density (compaction and baling); rate of water application controllable through the permeability of cover soils or synthetic membranes; and landfill depth or lift height.

Simulated landfill studies by Hentrich, et al. (1979) and Kemper and Smith (1981) indicate that shredding and baling of refuse have significant effects on contaminant flux and stabilization rates which are amenable to the following control measures:

- Baled refuse may be employed to enhance leachate production both by decreasing the time period before continuous production of leachate, and by increasing the overall volume of leachate produced. Baling results in large volumes of dilute leachate with a long period of time required for stabilization of the landfill;
- Shredding may be employed to promote rapid landfill stabilization through the rapid generation of concentrated leachate; and
- Shredding and baling both may be employed when the slowest possible rate of leaching and longest period of landfill stabilization are desired.

The amount of leachate generated in a landfill is directly related to the amount of water that infiltrates the fill. In general, the rates of biological and chemical stabilization in a landfill increase with the amount of moisture present until an upper limit is reached. Leckie and Pacey (1975; and Pohland

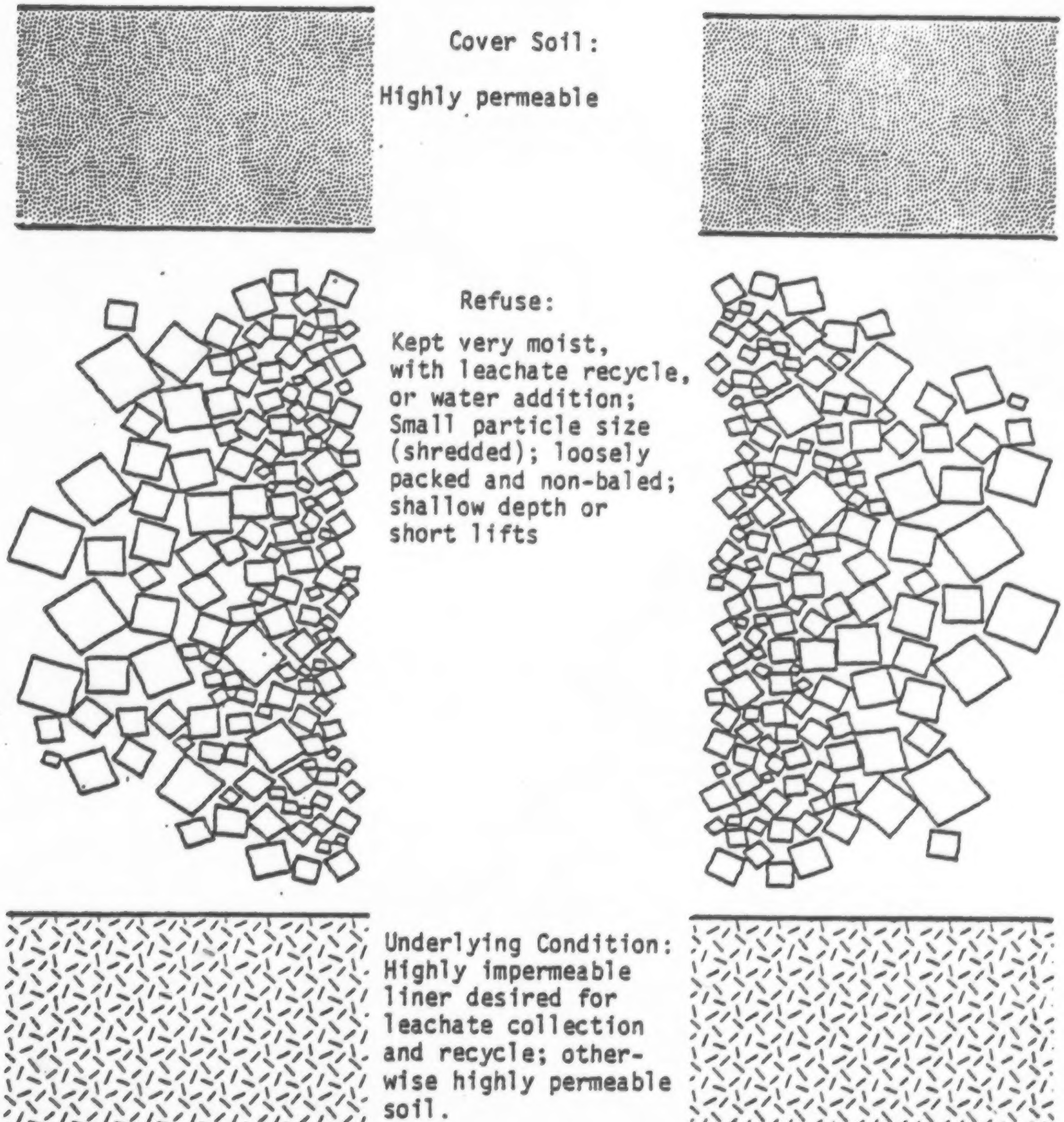
and Gould (1980) have established that water application techniques are advantageous to accelerated landfill stabilization, i.e., leachate production. Depending on whether rapid or gradual landfill stabilization is desired, surface water infiltration may be increased by the installation of a highly permeable cover soil; or reduced by landfill contour grading, diversion ditches, and/or relatively impermeable surface seals.

Stabilization rates generally increase with temperature until an upper limit is reached. Temperature is usually not a restricting factor in northern latitudes since the interior of the refuse mass is insulated from the ambient temperature and the degradation process is exothermic in character (Zononi, 1972). It is concluded that regulation of the refuse temperature to decrease or increase stabilization rates would not be an easily implemented control technique. Available moisture then becomes the limiting factor.

The influence of refuse depth on leachate composition is a factor of the effective contact time between percolate and refuse. Column tests by Fungaroli, et al. (1979) showed that upper levels of refuse stabilize earlier than deeper levels. This behavior indicates that shallow landfills will reach stabilization much earlier than deeper ones; consequently, the release of contaminants will be distributed over a longer period for deeper fills (Qasium and Burchinal, 1970). These studies indicate that the regulation of landfill depth may be a viable method for controlling landfill stabilization rates; however, the economics of this approach has not been established.

Two distinct refuse stabilization rates, resulting in two distinct kinds of leachate and contaminant flux regimes are recognized through manipulation of the above-mentioned construction variables: (1) rapid refuse stabilization, conducive to rapid contaminant extraction per unit time and, ultimately, a relatively dilute leachate if adequate soil attenuation capacity is available, and (2) gradual refuse stabilization, conducive to suppressed contaminant release per unit time and a maintenance of concentrated leachate for extended periods. Each of these two conditions and corresponding control measures governing the appropriate leachate response are presented in Figures 47 and 48. The leaching effects in each case are realized only after a landfill has attained field capacity. Leachates from landfills which have not reached field capacity are largely controlled by conditions of permeability (including channeling effects) which can greatly alter leachate trends.

Two points of view are expressed in the literature as to whether a landfill site should be designed for gradual or rapid stabilization. One viewpoint is that if the stabilization process is retraded by restricting the infiltration of water, more time will be allowed for the natural attenuation process to take



Cover Soil:

Highly permeable

Refuse:

Kept very moist, with leachate recycle, or water addition; Small particle size (shredded); loosely packed and non-baled; shallow depth or short lifts

Underlying Condition: Highly impermeable liner desired for leachate collection and recycle; otherwise highly permeable soil.

Figure 47. Landfill design and operational features promoting rapid landfill stabilization.

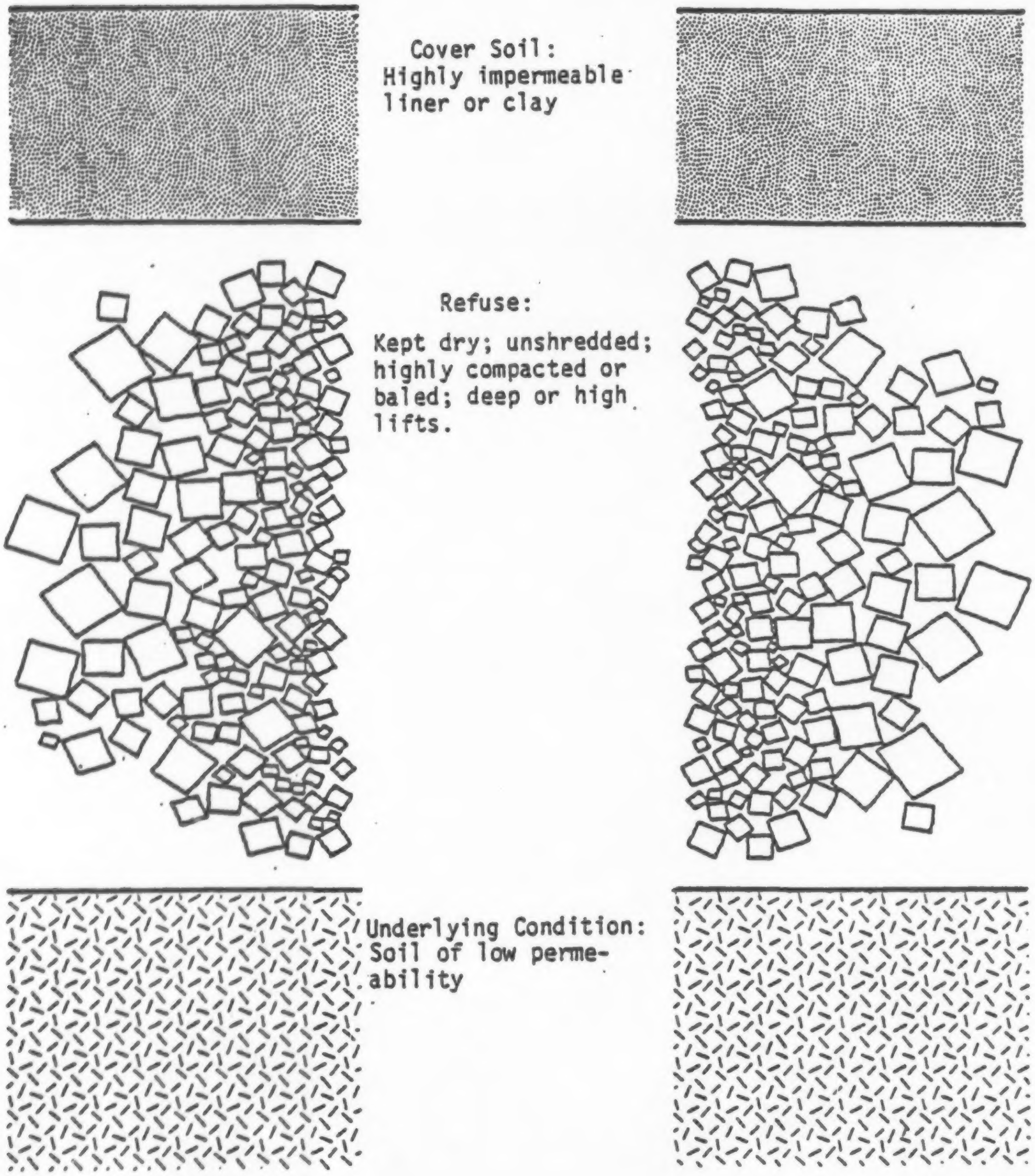

Cover Soil:
Highly impermeable liner or clay
Refuse:
Kept dry; unshredded; highly compacted or baled; deep or high lifts.
Underlying Condition: Soil of low permeability

Figure 48. Landfill design and operational features promoting gradual landfill stabilization.

place. For this scenario, it is argued that a combination of retarded degradation in a geologic formation of adequate attenuation provides the least likelihood of serious ground-water pollution. It should be noted that the gradual stabilization approach may require long term post-closure landfill monitoring and management. Advocates of this approach further declare that increasing the stabilization rate by maximizing the infiltration of surface water imposes a greater pollutional load in a shorter time span on the attenuation capability of the geologic formation which may result in significant degradation of the ground-water quality. It should be noted that the same potential pollutional load is involved in both cases.

The other viewpoint expressed by some is that a landfill should be designed for optimum stabilization by permitting surface water to percolate through the fill. For this approach, it may be necessary to collect and treat the leachate prior to discharge to a surface water body, or possibly, recycle the leachate through the fill. After a reasonable degree of stabilization has occurred, the leachates will no longer be collected but allowed to pass into the surrounding soil. In this way, it is argued that the stabilization process can be controlled as desired and the possibility of future pollution problems is reduced considerably. It is further argued that large quantities of landfill refuse are potential pollutional "time bombs" which could exist for many years after landfill closure. There is no question that landfill sites can be designed and operated to speed up the degradation process and collect and treat the leachate; however, the economics of this approach may favor other disposal methods.

Leachate Recirculation

The stabilization of landfill refuse and the quality of leachate are largely the result of biological processes. Leachate recirculation essentially uses the landfill as a generally uncontrolled anaerobic digestor for effective anaerobic treatment of its own leachate (Leckie, et al, 1975). Recirculation systems offer several advantages in terms of leachate control, including acceleration of landfill stabilization and quality control of the leachate. Also, treatment facilities are not usually required for collected leachate. Cited disadvantages are the high capital and maintenance costs, above-grade systems often result in odor problems, and the possible increase in leachate volume production resulting in increased handling costs (Tolman, et al., 1978).

Three types of leachate recirculation systems are recognized: spray irrigation, overland flow, and subgrade irrigation. A summary of these systems and leachate collection systems is given in Table 65.

TABLE 65. LEACHATE RECIRCULATION AND COLLECTION TECHNOLOGIES (Tolman, et al., 1978)

Control Techniques	Optimum Site Conditions	Limitiations	1981 Costs	Perceived Effectiveness
Leachate Recirculation				
Spray Irrigation	High evaportation rates for the site; site vegetation well established. Site contour minimizes ponding, runoff, etc.	Practical only in dry months; wet weather conditions minimizes effectiveness. Pumping equipment, collection wells and associated piping is required.	Capital Costs $21.30/lineal ft. Operating and maintenance costs $2.40/lineal ft/yr. Power costs $16.30/lineal ft/yr.	Most effective in arid regions or during the summer months when treatment efficiency is the highest.
Overland Flow	Land availability with existing leachate collection, spray equipment, and ditches for recycle; existence of rooted grasses and slope conducive to natural gravity flow. Favorable soil properties and high evaporation/evapotranspiration rates desired.	Large area requirements; leachate collection and distribution system required. Conducive weather conditons required for optimum use.	Capital Costs $3.20/lineal ft. Operating and maintenance costs $1.73/lineal ft/yr. Power costs†	Provides an effective treatment scheme when land is available and climatic conditions are favorable.
Subgrade Irrigation	Collection and leachate distribution systems exists for the fill.	Elaborate piping and distribution system required.	Capital costs $7.30/lineal ft Operating and maintenance costs $1.07/lineal ft/yr. Power costs†	Provides an effective landfill stabilization mechanism. Largely unaffected by climatic conditions. Treatment efficiency is less than for spray irrigation or overland flow.
Leachate Collection Systems				
Shallow Drains	Shallow bedrock condition; homogeneous off-site soils.	For most effective use, require installation prior to landfilling. Difficult to repair once damaged. Subject to clogging.	Capital costs $3.40 to $4.85/lineal ft. Operating and maintenance costs $1.36/lineal ft/yr.	Provides an effective collection method when used in conjunction with synthetic liners or bottom sealers.
Shallow Well Point System	Adequate separation of landfill and ground water.	Pumping costs are a major expense. May accelerate rate of leachate migration from the fill.	Capital costs $13.20 to $14.40/lineal ft. Operating and maintenance costs $7.30/lineal ft/yr. Power costs $16.70/lineal ft/yr.	Provides an effective method for collection of leachate downgradient of the land fill.

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* 1 ft = 0.31 m

† Power costs not available

Spray irrigation is accomplished by periodically pumping collected effluent through spray nozzles placed at 15- to 30-m (50 to 100 ft) intervals on the landfill. Spray irrigation has the advantage that some leachate treatment may be effected during spraying as a result of both aeration and infiltration through the cover soil.

Overland flow irrigation is an inexpensive well-developed technique by which trenches, spreading basins, or grated pipes are used to distribute the effluent. Leachate is periodically pumped into the distribution system and allowed to infiltrate into the ground. With both spray irrigation and overland irrigation, an impermeably lined pond is necessary to provide storage of the leachate between irrigation periods. No leachate overland flow programs are reported in the literature, although use of the treatment processes inherent in overland flow irrigation, represent a viable means of treating recycled leachate.

Subgrade irrigation has been postulated as a potential means to promote uniform stabilization of the fill. In these approaches, leachate is distributed throughout the fill through smaller subgrade tile fields or pipes. Very large tile fields must be subdivided into smaller units and each unit fed leachate separately. Subgrade irrigation may be used continuously and has the advantage of avoiding local odor problems.

Based on capital costs given by Tolman, et al. (1978) it is estimated that spray irrigation, overland flow, and subgrade irrigation systems would cost, respectively, $70.00 per m ($21.30 per lineal ft); $10.40 per m ($3.20 per lineal ft); and $24.50 per m ($7.46 per lineal ft), based on 1981 dollars. Operation, maintenance, and power costs for spray irrigation are estimated to be $63.40 per m ($19.30 per lineal ft) per year. Power costs for this system represent the major expense, i.e., $55.00 per m ($16.90 per lineal ft) per year. Operation and maintenance costs for overland flow and subgrade irrigation systems are estimated to be $5.67 per m/$1.73 per lineal ft) and $3.51 per m ($1.07 per lineal ft) per year, respectively. Annual power costs are not available for overland flow and subgrade irrigation systems.

Addition of Municipal and Industrial Sludges

The effects of adding municipal and industrial sludges to municipal refuse landfills are detailed in Section 4, entitled Leachate Composition. Evidence suggests that certain additions can be beneficial in accelerating the stabilization of a landfill (i.e., the concentration of organic pollutants and organometallic species appearing in leachate are reduced in several months rather than years). Research has indicated that receipt of certain alkaline or nutrient bearing wastes may promote a similar leachate response. An approach utilizing natural and synthetic materials as a bottom layer of a landfill has been demonstrated as a leachate composition control measure. These techniques are summarized as follows.

Municipal Wastewater Treatment Plant Sludge--

Sewage sludge may be added to municipal refuse for several reasons. The most common is to provide disposal by incorporating it with municipal refuse in a landfill. An auxilliary purpose of adding sewage sludge to refuse, however, may be to change the degradation processes within the landfill. It is the latter application which is of interest from the standpoint of leachate composition control.

Co-disposal of dewatered sewage sludge with municipal refuse is a common practice at multipurpose landfills. Studies by Stone (1973), Emcon Associates (1974), and Pohland and Gould (1980) have indicated that sewage sludge can act as a buffering agent, thereby, maintaining a landfill pH at levels (> 6.0) which can increase stabilization. Additionally, the sludge can enhance refuse decomposition by possessing a high microbial population and high moisture content. There is concern, however, that addition of sewage sludge may result in pathogens, and toxic organics and trace metals in the leachates and will pose a threat to ground-water supplies.

One method of co-disposal involves the placement of refuse at the top of the working face along with the sludge. The two wastes are mixed thoroughly and spread thinly and compacted upon the working face. Techniques have been developed to dewater and solidify the semi-solid sludge for conversion into a granular solid suitable for landfilling. High costs will likely prevent the use of this co-disposal method.

Industrial Wastes--

A distinct lack of data is evident for the technologies associated with landfill disposal of industrial wastes. Little is known on what effect adding industrial sludge to municipal refuse has on the stabilization process and the quantity and quality of leachate produced during decomposition. There is a strong concern that addition of industrial wastes, particularly those high in heavy metals, will result in elevated metal concentrations in the leachates and potentially, in potable ground-water supplies. Advocates of co-disposal of sludges and municipal waste believe the presence of organics in the landfill will immobilize heavy metals. They also believe the presence of such sludges may accelerate the decomposition process and shorten the time required for biological stabilization of the refuse (Shoemaker and Rittenhouse, 1980).

Leachate stabilization is usually accelerated in a landfill environment in which the pH is low, the chemical environment has become reduced and a primary determinent of the presence or absence of various constituents will be their relative solubilities in the leachate medium. The availability of many heavy metal species (e.g., Zn^{+2}, Pb^{+2}, Cd^{+2}, Cu^{+2}) is likely governed by the presence of sulfide, which in most cases, exists in concentrations sufficient for precipitation. Under reducing conditions and an increasing pH consequenced by the conversion of volatile acids to methane in a biochemically active landfill, metal sulfide solubilities become even lower. Thus, the co-disposal of heavy metal-laden industrial wastes with municipal refuse in a controlled manner may mitigate the environmental impact of such disposal practices (Pohland and Gould, 1980).

Pilot-scale investigations by Pohland and Gould (1980) indicate that significant attenuation and reduction of contaminant concentrations (particularly of heavy metals) in leachate from landfills receiving industrial waste is possible provided appropriate landfill conditions exist (i.e., reducing conditions, high sulfide concentrations).

Among the environmental factors required for support of active biological stabilization of leachate constituents are the availability of substrate and nutrients. Leachate data reported by Pohland and Gould (1981), indicate that phosphorus may be limiting in a landfill environment. This premise tends to be substantiated by the indicated BOD_5:N:P ratios of about 100:10:0.3 in leachate from a pilot-scale landfill cell. Thus, additions of wastes with phosphorous in the proper chemical state may prove beneficial in restoring stabilization.

Because of the high moisture content and the high alkalinity of many industrial sludges, the inclusion of alkaline wastes may prove beneficial in accelerating refuse stabilization. By increasing leachate pH to levels which enhance methane fermentation, levels of leachate organics may be diminished. Additional research is required to define the applicability and provide quantitative information to optimize this procedure.

Addition of Selected Sorbents

Laboratory studies by Liskowitz, *et al*. (1976) have demonstrated the effectiveness of various natural and synthetic materials in removing contaminants from leachates. Selected materials included bottom ash, fly ash, vermiculite, illite, ottowa sand, activated carbon, kaolinite, natural zeolities, activated alumina, and cullite. This study showed that the behavior of a sorbent as regards the removal of a particular contaminant, varied with the type of leachate being treated. This effect may result from the competitive ion exchange and sorption processes which exist among the constituents of a leachate and the sorbent involved.

The results reported by Liskowitz, et al. (1976) indicate that no single sorbent, of those examined, can significantly reduce the concentration of all constituents in leachate to acceptable levels. The results, however, suggest that combinations of different sorbents can be used to reduce most of the contaminants in the leachates to acceptable levels. The most promising use of sorbents are for leachate treatment following collection; however, placement in landfills as a bottom layer to provide substantial contaminant reduction in leachates prior to migration from the sites, may exhibit greater economical feasibility due to the reduced contaminant load imposed upon the leachate treatment facilities.

Studies by Fuller and Artiola (1978) indicate that the use of limestone as an additional attenuation layer on top of natural or synthetic layers has the ability to alter the pH of a leachate solution, thereby, effecting pollutant removal. Crushed limestone has proven effective as a low-cost aid in the migration control of certain heavy metals. The limestone apparently effects the mobility of the hazardous constituents of the leachate by: (1) absorbing the metallic ion directly; (2) forming less soluble calcium and carbonate compounds; and/or (3) raising the pH of the particular leachate as it passes through the limestone. Practical use of crushed limestone as liner material for landfills as an aid in reducing the migration rate of potentially toxic metal pollutants from leachates is suggested (Chen and Eichenberger, 1981).

LEACHATE COLLECTION SYSTEMS

Shallow interceptor drains and well systems are most commonly used for leachate collection. The equipment and techniques are identical to that described for leachate volume control, though leachate rather than indigenous waters are handled. See Table 65 for a summary of leachate collection technology and costs.

Shallow Drains

Drains may be employed to intercept leachate from a shallow ground-water table such as in the case of trench and area fill landfill operations. Drains may also be used to collect leachate along a steep hill downgradient from a disposal area.

Lutton, et al. (1979) and Lutton (1980) cite the use of an underground drainage system incorporated into a cover seal for a sanitary landfill. Construction, methods, and basic costs of drains are identical to those used to intercept ground water prior to entering a disposal site.

Steiner, et al. (1979) described a sanitary landfill collection system in Pennsylvania which allowed the leachate to collect by gravity at three locations equipped with manholes.

Leachate was pumped from these manholes and to the treatment facility via pressure lines.

Well Systems

Shallow well point systems located down-gradient from a landfill, or a deeper well system utilizing an underdrain collection system, have been implemented at sanitary landfills throughout the U.S. The model Lycoming County landfill utilized a performated pipe network installed within a 15.2 cm (6 in.) sand filter for leachate collection above a membrane (Giddings, 1977). A second drain collection network was installed under the membrane to protect ground waters from leachates leaving the landfill via membrane leaks. Similar systems have been described for which leachate was collected at contoured low points of the landfill bottom through collection wells (Mang and Weaver, 1976).

LEACHATE TREATMENT

Control systems which involve the interception and collection of municipal landfill leachates may also require treatment of the collected leachate prior to ultimate disposal. Leachate treatment has been the subject of considerable research over the past decade, the results of which are discussed below.

Section 4, Leachate Composition, describes the chemical and physical character of municipal landfill leachate. A cursory review indicates that, in general, fresh leachates are characterized by very high BOD and COD concentrations (i.e., in the range of 10,000-50,000 mg/l or higher). These concentrations, however, are likely to decrease with continued leaching and biostabilization. High concentrations of major cations and anions, and suspended solids are also present near the onset of leachating, and similarly tend to decrease as percolation through the solid waste continues. The presence of significant concentrations of metals, including copper, cadmium, lead, mercury, and zinc, are of particular environmental and public health concern. Unlike the organic and major ionic concentrations of leachate, heavy metal concentrations are generally unpredictable as leaching continues. Nitrogen and phosphorus may be present in leachate though at levels below that required for uninhibited biological utilization of the organic matter present. Although the acidity or alkalinity of leachate varies, organic acids formed by acid-forming bacteria during anaerobic decomposition of solid waste may create an environment that is slightly acid.

In addition to the long-term changes in the composition of leachates mentioned above, short-term variability of specific components is observed (Steiner, et al., 1979). These short-term changes are produced by seasonal effects such as increased infiltration or evapotranspiration, or through hydrologic events.

The combined effects of short- and long-term leachate variability makes the design of a treatment system particularly difficult.

The aforementioned information indicates that municipal landfill leachate can be quite different from domestic wastewaters, and thus may not be treatable through conventional municipal wastewater treatment systems. Leachates may have a much higher organic content than wastewater, as well as generally higher concentrations of toxic substances such as metals. Also, whereas a particular domestic wastewater may exhibit stable concentrations over time, leachates can be highly variable. Further evaluation of the potential for leachate treatment by conventional wastewater treatment plants is provided later in the discussion.

Basic Leachate Treatment Methods

Leachate treatment techniques demonstrated in recent years are comprised of the traditional treatment processes common to wastewater systems. The suggested general approaches to treatment are:

- Biological treatment-aerobic and anaerobic biostabilization systems; and
- Physical and chemical treatment-precipitation, adsorption, coagulation, chemical oxidation, and reverse osmosis.

After an extensive literature survey of treatment methods for landfill leachates, Chian and DeWalle (1976) concluded that newly formed landfill leachate is best treated by biological treatment processes. Leachate from stabilized landfills, i.e., leachates of low organic content, are best treated with physical and chemical methods.

Biological Treatment of Leachate--

Various researchers have investigated the potential of aerobic and anaerobic processes for the stabilization of leachate. The primary goal of these approaches is to reduce high BOD and COD concentrations in leachate. A basic problem in biological treatment is that the leachate metals and other contaminants may exert toxic effects on the biological treatment culture. Evaluation of the kinetic factors in leachate treatment exhibits these inhibitory effects.

Aerobic treatment--Laboratory-scale research has demonstrated that activated sludge processes can effectively remove organic matter and metals from leachate (Boyle and Ham, 1974; Cook and Foree, 1974; Uloth and Mavinic, 1977; Palit and Qasim, 1977). Removal of 90 to 99% of the leachate BOD and COD and some metal removal, was commonly reported. The MLVSS concentration in these lab-scale reactors was higher than that typically observed in wastewater treatment: (e.g., 5,000 to 10,000 mg/l for leachates compared to 1,000 to 2,000 mg/l for typical municipal wastewater processes). Because of the high suspended solids concentrations, the food to microorganism ratio was low (0.02-0.06). Hydraulic detention times are substantially higher for leachates then for municipal wastewater (e.g., 1 to 10 days for leachate compared to several hours for wastewater). Solids detention times were 30 to 60 days for leachate compared to 5 to 15 days for municipal wastewater. These operational characteristics indicate that the large amount of organic matter in leachate is not readily oxidized, and requires extensive biological activity for stabilization. The long treatment times indicate that extensive aeration energy requirements will be required for aerobic treatment of leachate.

Various other operational problems were observed during aerobic treatment, including foaming, nutrient deficiencies, and toxic inhibition. Uloth and Mavinic (1977) indicated that excessive aeration in conjunction with high concentrations of metals contributed to foaming, and that mechanical mixing independent of aeration and anti-foaming admixtures, could help obviate this problem. Palit and Qasim (1977) indicated that leachate stabilization could be hampered by nutrient deficiences, and that the addition of nutrients may be necessary in some cases. The toxic effects of metals and other constituents appear to inhibit biological removal of oxygen demanding material, as indicated by the increased time required for biostabilization (Uloth and Mavinic, 1977).

With regard to stabilization kinetics, Uloth and Mavinic (1977) evaluated the kinetics of aerobic treatment of high strength leachate in comparison with municipal wastewaters. Table 66 shows the kinetic values when evaluated according to the Lawrence and McCarty (1969) Model. The values shown in Table 66 indicate a subdued level of microbial activity in the treatment of leachates in comparison to the treatment of municipal wastewaters.

Metals removed from leachate during aerobic treatment was reported by Uloth and Mavinic (1977). Following a biological detention time of 10 days, activated sludge digester effluents showed less than 10 mg/l of iron, dropping from an original concentration in raw leachate of 240 mg/l. More than 95% of the mixed liquor aluminum, cadmium, calcium, chromium, manganese, and zinc were also removed by the settling biological floc. An average of 85% of the lead and 76% of the nickel was associated with the sludge solids. Between 49% and 69% of the mixed liquor magnesium was removed by settling. The authors noted that the percentage removal for all metals is generally higher than that observed by other researchers for activated sludge processes.

TABLE 66. KINETIC PARAMETERS FOR AEROBIC TREATMENT OF LEACHATE (HIGH STRENGTH) AND DOMESTIC WASTEWATER (Uloth and Mavinic, 1977)

	Y	k_o Day-1	k_m mg/l	k_d Day^{-1}
High Strength Leachate	0.322	0.75	21375	0.0025
Typical Municipal Wastewater	0.6	6.0	100	0.05

Y = growth yield coefficient (biological solids produced/ COD destroyed) lb/lb

k_o = maximum rate of substrate utilization per unit weight of biological solids

k_d = endogenous death rate of biological solids

k_m = substrate concentration when specific substrate utilization is one half the maximum

Anaerobic treatment--Anaerobic treatment of leachates has been effective in reducing organic loads. Continuous culture lab-scale reactors have demonstrated 90 to 99% organic removal which is comparable to that achieved by aerobic treatment (Boyle and Ham, 1974; Pohland, 1975). Similar removal efficiency is reported by Chian and DeWalle (1976) using a lab-scale anaerobic filter.

Anaerobic treatment offers several advantages over aerobic treatment. First, the build-up of biological solids in the reactor is low, implying a reduction in sludge disposal requirements. Second, treatment times demonstrated for anaerobic treatment of leachate (about 10 to 15 days) are comparable to the time required for anaerobic digestion of municipal waste water. Finally, the absence of aeration requirements allows a saving of power costs, though some energy may be required to maintain reactor temperatures at the necessary 30-35°C. A major product of anaerobic digestion is methane, which can be used as an energy source for maintaining reactor temperatures, or for other purposes.

Physical/Chemical Treatment of Leachate--

Various physical/chemical treatment processes have been applied to leachate treatment. Physical or chemical treatment processes may be particularly useful in treating leachates from older landfills whose organic content is negligible, or as a polishing step for leachates previously treated by biological methods. Some of the major physical/chemical treatment processes demonstrated are described below.

Precipitation and coagulation--Chemical precipitation and coagulation experiments have been fairly successful in removing iron, color, and suspended solids in leachates. Ho, et al. (1974) and Thornton and Blanc (1973) have demonstrated that precipitation with lime is effective in the removal of iron and other multivalent ions, color, suspended solids, and COD. BOD concentrations, however, are apparently unaffected. Iron precipitation by lime is particularly pronounced, as nearly 100 percent removal is consistently reported at lime concentrations over 300 mg/l. Precipitation by sodium sulfide is reported by Ho, et al. (1974). Iron removal occurred only at a very high chemical dose (1000 mg/l), and its effects on other constituents was negligible. Sulfides are apparently not as promising for leachate cation removal as lime.

Coagulation with alum or ferric chloride is reported by each of the aforementioned researchers. Although some removal of iron and color was demonstrated, both coagulants were found to be of limited value in removing COD, chloride, hardness and total solids. Results were highly pH dependent, and chemical dosages were high, resulting in large amounts of solids production.

Ion exchange--Leachate treatment by ion exchange is reported by Pohland (1975) for leachates which had been previously treated biologically. Pohland reported good results using a combination of cationic and anionic exchange resins, removing many ionic species as well as dissolved solids and nutrients. Very little residual organic removal was reported; however, the use of mixed resins appears promising as a treatment approach for the nonorganic fraction of leachate.

Carbon adsorption--The removal of leachate organics by carbon adsorption was studied by Pohland (1975), for leachates previously treated by ion exchange. Though moderately successful, activated carbon was found to release solids which adversely affected the total solids concentration of the leachate. Pohland suggested that if leachate were to be treated with both carbon and mixed resins, the carbon treatment should precede the mixed resins. Activated carbon column tests performed by Ho, et al. (1974) demonstrated complete color and odor removals at a detention time of 20 minutes, along with 55 percent COD removal.

Reverse osmosis--Chian and DeWalle (1976) examined reverse osmosis treatment of leachates and found effective removal of COD and dissolved solids. However, the potential for membrane fouling by suspended solids, colloidal material, and iron hydroxides, was noted. Thus, reverse osmosis is perhaps most effective as a post-biological treatment step for removal of residual COD and dissolved solids. Chian and DeWalle also note that membrane efficiency is sensitive to pH.

Chemical oxidation--Chemical oxidation was shown to be reasonable effective in removal of COD, iron and color (Ho , et al. 1974), though only at high concentrations of the oxidizing agent. Oxidizing agents included chlorine, calcium hypochlorite, potassium permanganate, and ozone.

As a means of summarizing the treatabilities of landfill leachates, Figures 49 and 50 present the removal efficiencies of the various treatment processes reported in the literature. Figure 49 portrays the COD removal efficiencies for raw leachates of a young landfill (or fresh refuse), and the COD removal efficiencies following initial biological treatment. Each process is subject to a number of design variables, which may in part, account for the range of observed efficiencies. According to Figure 49, the anaerobic filter, anaerobic digestion, and activated sludge processes, show the greatest COD removal efficiency for raw, non-biologically treated leachate. For COD removal following initial biological treatment, activated carbon and reverse osmosis treatment approaches provide the highest efficiency. Coagulation and oxidation systems generally remove less than 40 percent of the leachate organics. Figure 50 presents the reported removal efficiencies for various inorganic contaminants by aforementioned unit treatment processes. A comprehensive summary of the effectiveness of chemical oxidation, chemical precipitation, ion exchange, and reverse osmosis treatments of sanitary landfill leachate is provided by Shuckrow, Pajak, and Touhill (1981).

Leachate Treatment Systems

Because leachates can have high concentrations of both organic and inorganic contaminants, treatment of leachates at the field scale requires integration of the basic treatment methods into a systematic approach. The design of such a system should account not only for the volume and quality of leachate to be treated, but also on the changes in leachate quality over time. Several leachate treatment system options are: complete treatment of leachate, pretreatment or combined treatment with municipal wastewater, land application of leachate, aerated lagoon, and leachate recirculation through the landfill. The following text describes these basic treatment systems.

Complete treatment of leachates refers to some sort of combination of the basic treatment schemes discussed earlier. The complete treatment approach is perhaps best suited for landfills remotely located from an existing sewage treatment plant. Steiner et al. (1979) demonstrated a complete treatment system

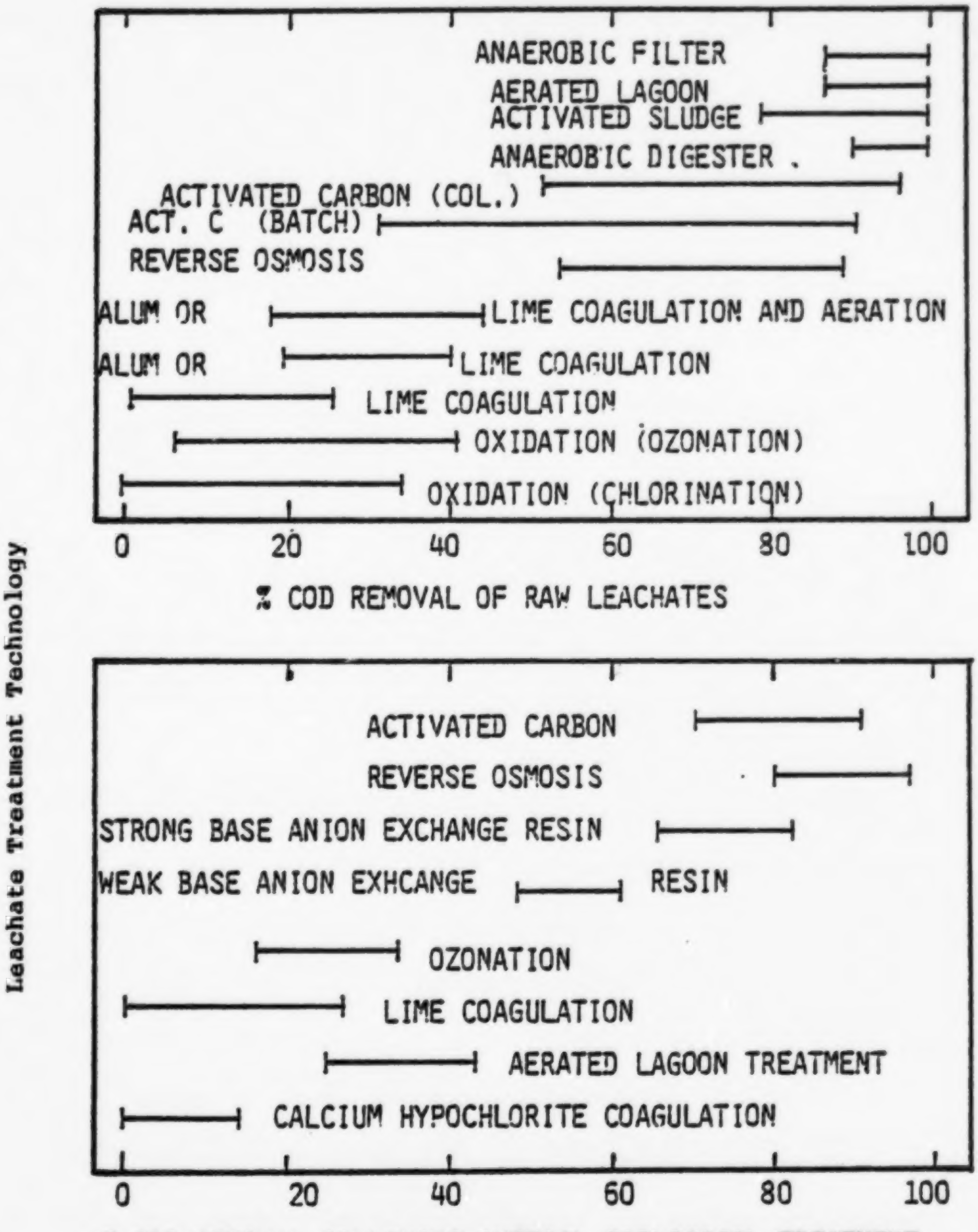

Figure 49. COD removal efficiencies for the treatment of leachates (Boyle and Ham, 1974; Cook and Foree, 1974; Ho *et al*., 1974; Palit and Qasim, 1977; Uloth and Mavinic, 1977; Steiner *et al*., 1979; Chian and DeWalle, 1977).

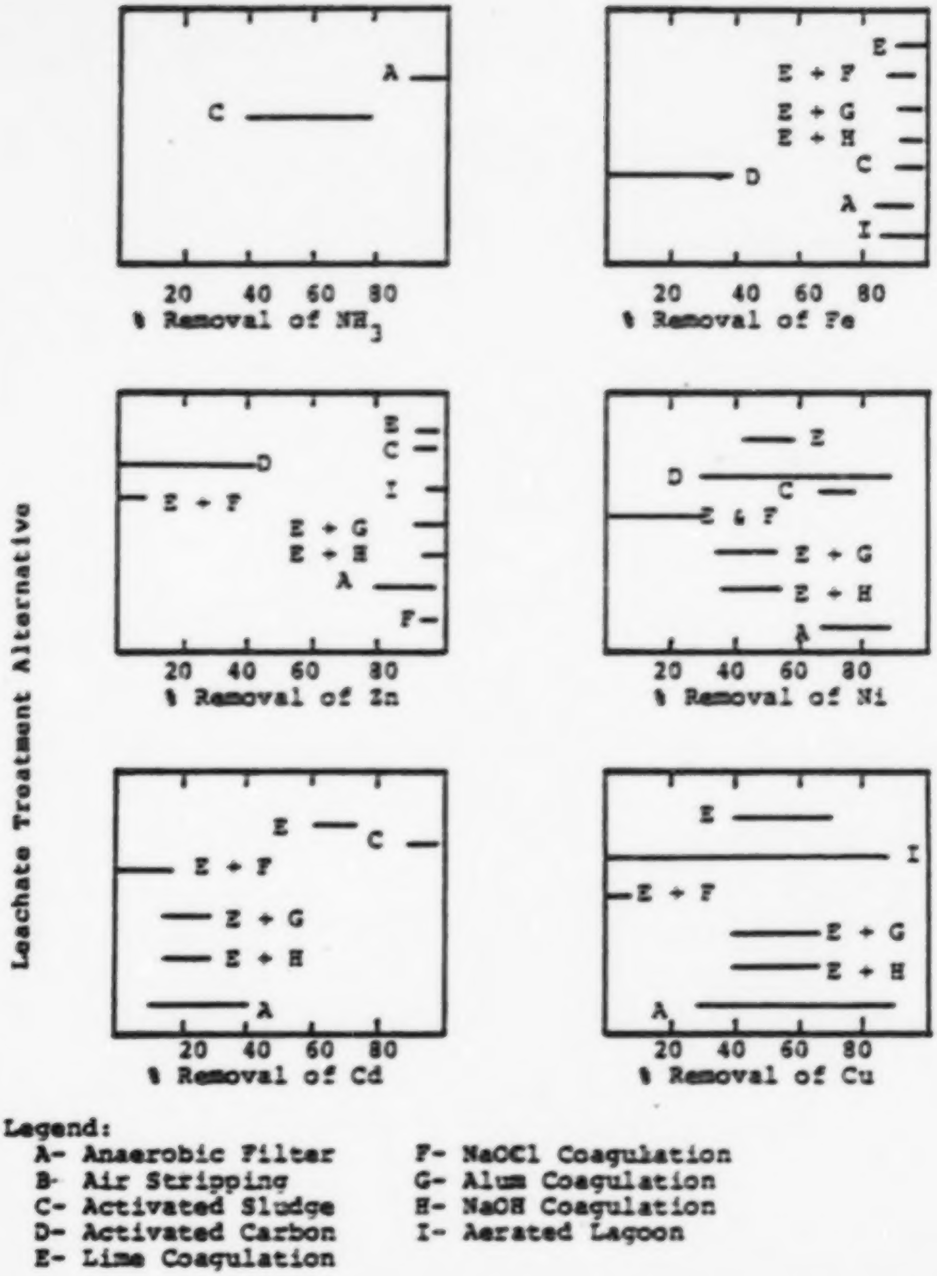

Figure 50. % removal of NH_3, Fe, Zn, Ni, Cd, and Cu by various treatment alternatives (Thornton, and Blanc, 1973; Boyle and Ham, 1974; Cook and Foree, 1974; Ho, *et al.*, 1974; Pohlnad, 1975; Chian and DeWalle, 1977; Uloth and Navinic, 1977; and Steiner, *et al.*, 1979).

utilizing physical, chemical, and biological unit processes. Leachate produced by a landfill in Tullytown, Pennyslvania is collected by an asphalt liner. Several unit treatment process sequences have been tried. The best treatment sequence, and the only one that is capable of approaching effluent standards, provides flow equalization, lime precipitation and clarification, ammonia stripping, activated sludge, and chlorination. The design is illustrated in Figure 51. A summary of the plant operating data appears in Table 67.

TABLE 67. SUMMARY OF SYSTEM 1 OPERATION DATA (Steiner, 1979)

Parameter	Raw Leachate mg/liter	Final Effluent mg/liter	Percent Removal	Discharge Standard mg/liter
Ammonia-N	758	75	90.1	35
BOD_5	11886	153	98.7	100
Cadmium	0.08	0.017	78.2	0.02
Chromium	0.26	0.07	73.1	0.1
COD	18490	945	94.9	*
Copper	0.40	0.11	72.5	0.2
Iron	333	2.7	99.2	7.0
Lead	0.74	0.12	83.8	0.1
Mercury	0.006	0.004	27.4	0.01
Nickel	1.76	0.75	57.4	*
Zinc	19.5	0.53	97.3	0.6

* No discharge standard for this parameter

Because of low flow conditions of the receiving waters, treated leachate can be discharged only from December until April. In other months, treated leachate is recycled to the landfill. Leachate to be recycled is not chlorinated.

Major problems reported for this operation include leachate flow and strength variability. Leachate COD has varied over a wide range from day to day, though concentrations averaged between 10,000 and 15,000 mg/l during the first three years of operation. A lime precipitation stage was needed to reduce the metals concentrations to levels which would not inhibit the

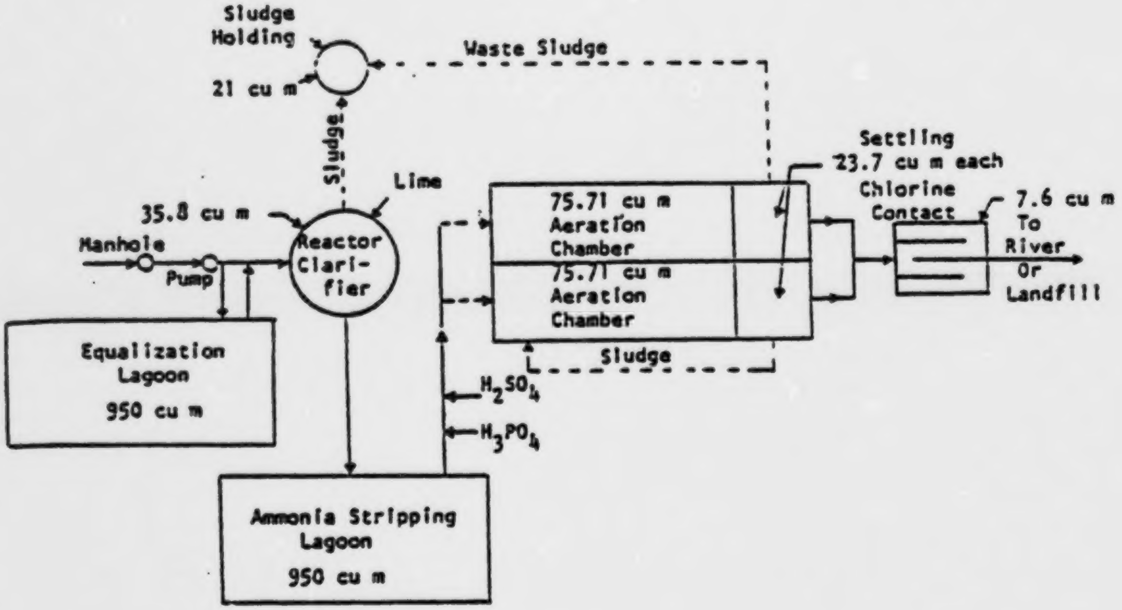
Sludge
Holding
21 cu m
Waste Sludge
Sludge
Lime
35.8 cu m
Manhole
Pump
Reactor
Clari-
fier
Equalization
Lagoon
950 cu m
Ammonia Stripping
Lagoon
950 cu m
H_2SO_4
H_3PO_4
75.71 cu m
Aeration
Chamber
75.71 cu m
Aeration
Chamber
Sludge
Settling
23.7 cu m each
Chlorine
Contact
7.6 cu m
To
River
Or
Landfill

Figure 51. Schematic of complete leachate treatment plant (Steiner, et al., 1979).

activated sludge process. Similarly, the toxic effect of ammonia was controlled by the ammonia stripping process. After lime treatment, sulfuric and phosphoric acids are added to lower the pH and raise nutrient levels before the activated sludge process. The activated sludge process reduced BOD and COD concentrations by over 90 percent (Table 67), though a detention time of more than one day is typical.

Steiner, et al., report that the system operation is sensitive, and effluent standards are approached only during optimum operation of the plant. Thus, although overall satisfactory complete treatment of high strength leachate can be accomplished, it is clear that a sophisticated treatment system is needed. A comprehensive review of process train alternatives for hazardous waste leachates is presented by Shuckrow, Pajak, and Touhill (1981).

For leachate collected from a landfill located near a wastewater treatment system, a convenient method of treatment would be to discharge the leachate to the sewer system. The possibility of mixing untreated leachate with domestic wastewater for conventional biological treatment was studied by Boyle and Ham (1974). Their findings indicated that high strength sanitary landfill leachate (10,000 mg/l COD) can be added to domestic wastewater in an extended aeration activated sludge plant at a level of at least 5 percent by volume without seriously impairing the treatment process or the effluent quality. Beyond 5 percent by volume, leachate additions resulted in substantial solids production, increased oxygen uptake rates, and poorer mixed liquor separation. Chian and DeWalle (1977) also studied combined treatment, and found that leachate inflows greater than 4 percent reduced treatment efficiency. Boyle and Ham (1974) suggested that the presence of metal and ammonia toxins and extremely high organic loads in leachate could cause severe upsets in biological reactors.

Where toxins are expected to be a problem, or for leachate flows representing a large fraction of the domestic load, leachates may be pretreated prior to mixing with domestic flows. A simple pretreatment system involving an aerated basin with a detention time of 10 days, and a polishing basin with a 10-day detention time, was reported by Vydra and Grimm (1976). Actual performance data was not available for the facility. In the event that the pretreatment system can bring leachates to within effluent guidelines, the leachates will be discharged to an adjacent river rather than the treatment plant.

Land application of leachates through spray irrigation or overland flow is another possible treatment alternative. Since the soil is the media in which the treatment is to take place, careful consideration must be given to selecting a site with soil properties which are suitable for the retention and degradation of the leachate to be treated. A detailed soil survey conducted according to the standard Soil Conservation Service (SCS) technique is necessary for all land treatment facilities (K. W. Brown and Associates, 1980). A thorough knowledge of the hydrogeologic setting of the site is also required.

The possibility of land application of leachate from a sanitary landfill on a sandy soil in a high water table area in Manatee County, Florida was investigated by Nordstedt, et al. (1975). Collected leachates were pumped to a detention pond, where a sprinkler irrigation system discharged the leachate over pasture grasses. Examination of soil solutions collected at 1-, 2-, and 3-ft (20-, 61-, and 91-cm) under leachate irrigated plots after 6 months, revealed no unusual changes. The authors suggest that land application of high strength leachates in sandy, high water table areas is a promising method of treatment and disposal. Among the advantages of land application are that fluctuating flow rates and water quality may be easily accommodated, and that equipment used to spread the leachate may be moved to another location when a site is completed.

Although not considered a land application system, the migration and attenuation of leachate contaminants by soils underlying a landfill is similar to a land application system. The land disposal gudelines promulgated under RCRA (EPA, 1979) suggest full assessment and subsequent utilization of the hydrogeologic capability of subsurface soils to attenuate leachate pollutants as a major leachate control alternative. Fuller (1978) performed a comprehensive investigation of leachate pollutant attenuation by soils, and suggests that soil attenuative ability depends largely on its clay and silt content, hydrous oxides of iron, and pH.

Aerated lagoon treatment of high strength leachate (COD = 57,900 mg/l) was examined by Chian and DeWalle (1977), who utilized completely-mixed, laboratory-scale plastic tanks to simulate a lagoon. The experiments showed that between 93% and 97% of the organic matter of the leachates could be removed at detention times ranging from 86 days to as low as 7 days. Nutrient addition was deemed essential to successfully maintain effluent organic matter at low levels, and to prevent a decrease in MLVSS and deterioration of sludge settling rates. High removal of heavy metals was observed, especially for iron (> 99.9%), zinc (99.9%), calcium (99.3%), and magnesium (75.9%).

Perhaps the most innovative and acclaimed method of leachate treatment is the recirculation of high strength leachate back to the landfill, so that it may again percolate through the refuse. By recirculating the leachate, the organic component of the leachate can be reduced by the biological communities active within the refuse mass. Using experimental landfill columns, Pohland (1975) found that when leachate was continuously collected and reapplied to the landfill surface, the organic load of the leachate decreased to a small fraction of its peak value (i.e., from 20,000 mg/l COD to less than 1,000 mg/l) in a period of just over a year. A continued investigation of the effects of leachate recycle using a pilot-scale fill produced similar results (Pohland, 1979). Figure 52 shows the results of Pohland's original study and results from one of the field scale cells operated by Leckie, et al. (1979). The COD concentration histories show somewhat similar trends, with the leachate recirculation cells showing rapidly declining COD concentrations in comparison to the non-recirculating control cells.

Recycling of leachate apparently accelerates the stabilization of the readily available organic material in the landfill and in leachate. Pohland (1975) suggests the recirculation provides for a more rapid development of an active anaerobic bacterial population of methane forming bacteria. The rate of removal of organic components was further enhanced by the initial addition of sewage sludge and by pH control.

In addition to rapid decline in leachate COD concentrations over time provided by recirculation, similar concentration histories were observed for BOD, TOC, volatile acids, phosphate, ammonia-nitrogen, and TDS. Reduction in nitrogen and phosphorus are not considered great enough to limit biological growth. Recirculated leachate remains high in inorganic material.

Specific advantages of leachate recirculation include not only a substantial reduction in leachate organic components, but also a delay in the starting time for leachate treatment, and a leachate volume reduction through evapotranspiration loss. Leachate treatment costs may also be substantially reduced by recirculation as described below.

Treatment Costs

Costs associated with leachate treatment include leachate collecton costs and the costs of actual treatment. Leachate collection costs include the design and construction of an impermeable barrier or liner beneath the landfill, shallow drains, well points, and pumping equipment involved in leachate collection. A review of the costs associated with collection systems and pumping was provided in Table 65. It is generally estimated that a liner and leachate collection system will increase 30 to 40 percent the costs of an equivalent landfill without collection (Baldwin, 1979).

Costs associated with leachate treatment are largely undocumented, primarily because few field-scale treatment systems have been constructed. Chian and DeWalle (1977) estimated the total costs of leachate treatment (excluding collection costs) for three promising biological processes, and for physical/

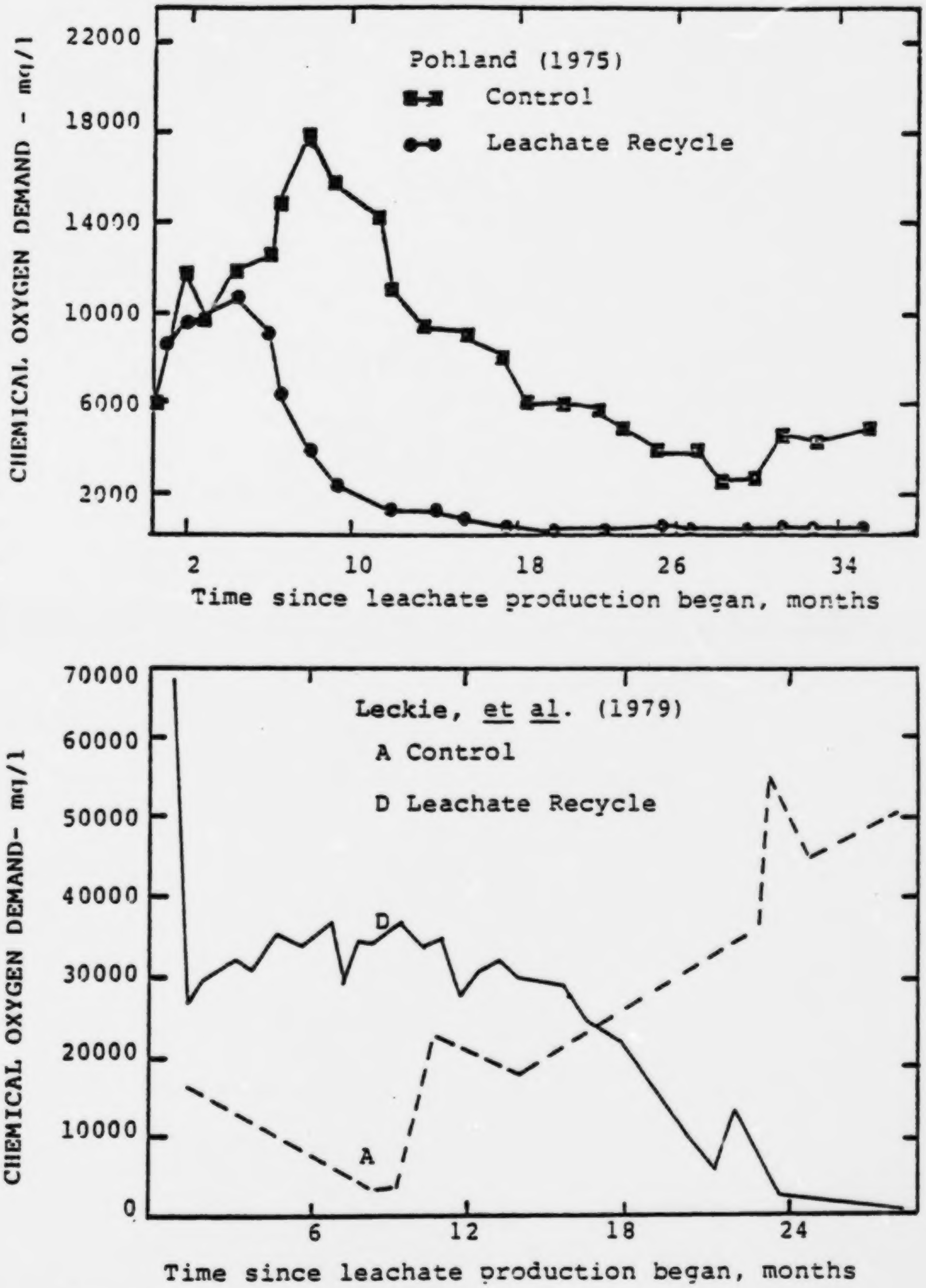

Figure 52. Effects of added moisture on leachate COD.

chemical treatment processes used to treat the effluents from the biological processes. The estimated costs are presented in Table 68 and are based on 1981 dollars. Table 68 shows that the treatment costs are a strong function of both leachate flow rates and leachate strength. In general, as the overall leachate strength increases, the unit cost of treatment increases greatly. Also, as the leachate flow rate increases, the unit cost of treatment decreases. The authors note that the circumstances of a particular situation may drastically alter treatment costs listed in Table 68. For example, for leachates discharged to a municipal wastewater treatment plant (activated sludge, combined treatment), having excess capacity the cost (in terms of surcharges paid) could be substantially lower than the estimates presented.

Steiner, et al. (1979) estimated the operating costs associated with the complete treatment system illustrated in Figure 51 to be approximately $4.20/1000 gallons, based on 1981 dollars. Power costs connected with the aeration of activated sludge and the ammonia stripping lagoon, amount to over 50 percent of this cost. Capital costs connected with the system were estimated by Straub (1979) using Engineering News Record and EPA cost estimation guidelines. Updated 1981 costs are:

Equalization lagoon	202,000
Time precipitation tank	144,000
Ammonia stripping lagoon	202,000
Activated sludge process	248,000
Disinfection tank	36,800
Total	832,800

Straub estimated total treatment costs of the system based on a 20 year design life. Making assumptions regarding the total volume of leachate generated over the 20 year period, energy costs, and an eight percent interest factor, Straub estimated that landfilling costs (on a cost/ton of refuse basis) are increased by 25-35% due to costs of collection treatment and handling of leachate.

Vydra and Grimm (1976) estimated the capital costs of the wastewater pretreatment system discussed earlier. The capital and construction costs of this system (including collection) was reported to be about 10 percent of the total construction cost of the landfill. Capital costs were kept low because of the use of native clay soils for the liner. The cost of installing an impermeable liner could more than double the capital cost figure. Operational costs were not reported.

For landfills where leachate collection and treatment is required, Straub (1980) indicated that leachate recirculation can substantially reduce treatment costs, regardless of the treatment system selected. The reduction stems from:

TABLE 68. A SUMMARY OF COST ESTIMATES FOR LEACHATE TREATMENT (Chian and DeWalle, 1977)

	Leachate* (gal/min)	Typical Effluent COD (mg/l)		1981 Costs ($/1000 gal leachate)*	
Influent BOD, mg/l		25,000	5,000	25,000	5,000
Activated Sludge (AS) (Combined treatment)	20	30	30	34.0	8.6
	2	30	30	59.6	17.1
Aerated Lagoon (AL)	20	500	100	25.8	5.9
	2	500	100	45.5	14.4
Anaerobic Filter (AF)	20	1500	300	31.8 † (25.8)	9.8 (8.5)
	2	1500	300	61.9 (55.9)	25.5 (24.2)
AL-Sand Filter (SF)-Activated Carbon (AC)	20	125	25	37.0	10.5
	2	125	25	57.5	19.7
AL-SF-AC-Reverse Osmosis (RO)‡	20	25	5	39.7	13.2
	2	25	5	64.2	26.5
AF-SF-AC	20	375	75	47.2 (41.2)	15.3 (14.0)
	2	375	75	78.0 (72.0)	31.7 (30.4)
AF-SF-AC-RO‡	20	75	15	50.0 (43.8)	18.0 (16.6)
	2	75	15	84.8 (78.2)	38.5 (36.6)

* 1 gal = 3.79 litre

† Numbers shown in parenthesis indicate the costs of treatment after deducting the credit for methane produced @ $1.50/1000 cu ft.

‡ After RO treatment the total dissolved solids (TDS) decreased to 300 mg/l and 60 mg/l for influent leachate BOD concentrations of 25,000 mg/l and 5,000 mg/l, respectively.

1. Internal reduction of the initially high organic load of the leachate; and

2. Reduction of the total volume of leachate due to increased evapotranspiration.

For a landfill operating without leachate recirculation, leachate COD can be extremely high at the onset of leaching (e.g., 25,000 mg/l or greater) and would diminish with continued leaching. The design of the leachate treatment system would need to account for the initially high organic load. Conversely, leachate drawn from a landfill with recirculation would have significantly lower organic concentrations. In addition, infiltrated precipitation from a landfill with recirculation would be subject to greater evapotranspiration as opposed to landfills without recirculation. The results of greater evapotranspiration is to reduce the total volume of leachate to be treated.

Comparative annual treatment costs for leachate from recirculated and non-recirculated landfills are presented in Table 69, based on the treatment system suggested by Chian and Dewalle, 1977, which includes an aerated lagoon, slow sand filtration, and activated carbon treatment. For each of the two hypothetical landfills, estimated treatment costs for landfills with recirculation is less than 25% of the estimated costs for the non-recirculated case. The major factor for this reported cost difference is the cost of treating the initially high COD present in the non-recirculated leachate (Straub, 1980). Note that the costs presented in Table 69 do not include the costs of a liner and collection system.

TABLE 69 . ESTIMATED COMPARATIVE ANNUAL LEACHATE TREATMENT COSTS FOR RECIRCULATED AND EQUIVALENT NON-RECIRCULATED LANDFILLS (Straub, 1980)

		1981 Costs	
Landfill area = 218,000m^2	**Treatment Volume m^3/yr**	**Cost/m^3**	**Cost/yr**
Recirculated Case	39,735	$2.28	$90,300
Non-recirculated Case	45,848	$8.83	$405,000
Landfill area = 22,500 m^2			
Recirculated Case	4,099	$4.52	$18,600
Non-recirculated Case	4,730	$15.72	$74,300

Landfills with leachate recirculation require a system for the reapplication of leachate to the landfill surface, which increase the costs of recirculation. By adding this cost to the figures in Table 70, the comparative savings per year in treatment of the recirculated leachate over the non-recirculated leachate is given as follows.

TABLE 70. COMPARATIVE SAVINGS OF RECIRCULATED CASE OVER NON-RECIRCULATED CASE (Straub, 1980)

		1981 Costs	
Landfill Area = 218,000 m^2	Treatment Cost/yr	Reapplication Cost/yr	Savings/yr for Recirculated Case
Recirculated Case	$90,300	$233,700	$80,500
Non-recirculated Case	$405,000	0	
Landfill Area = 22,500 m^2			
Recirculated Case	$18,600	$24,100	$31,700
Non-recirculated Case	$74,200	0	

For the larger landfill, a savings of 20% is estimated for leachate treatment and handling costs for the recirculated design over the non-recirculated design. For the smaller landfill, savings are estimated to be 43%. Thus, leachate recirculation becomes more attractive as the size of the landfill decreases, though savings are evident, regardless of size.

Leachate volume and quality estimates which serve as the basis for these cost estimates were obtained from the unsaturated flow model (Straub and Lynch, 1982b), described in the Leachate Composition section. The ability to predict leachate quality and quantity is invaluable in designing leachate treatment systems and in estimating the costs of such systems.

SUMMARY

Leachate volume control involves both ground-water and surface-water control measures. Subsurface infiltration barriers (e.g., bottom sealers, slurry trenches, grout curtains, sheet piling cutoff walls) are designed to either prevent ground water from flowing through the landfill and generating leachate or control the movement of leachate away from the fill. Drains, shallow well points, or deep wells are used to lower the water table to either prevent the formation of leachate or contain its spread. Surface water control measures such as contour grading and surface water diversion, surface sealing, and revegetation are used to minimize the quantity of water entering the landfill for purposes of reducing leachate generation.

The purpose of leachate composition control is to reduce the strength and contaminant flux of leachate. In general, high water application rates, small refuse particle size, and shallow landfill depths will increase refuse stabilization rates and, ultimately, the leachate strength. Leachate recirculation offers several advantages including acceleration of refuse stabilization, and treatment systems are not always required for the collected leachate. Codisposal of sewage sludge or alkaline wastes with municipal refuse can be beneficial in accelerating the stabilization of a landfill. Laboratory studies have demonstrated the effectiveness of various natural and synthetic materials in removing contaminants from leachate. The addition of crushed limestone as an additional attenuation layer has also proven effective as a low-cost aid in the migration control of certain heavy metals.

The successful treatment of high strength leachates will probably require different combinations of unit treatment techniques, the appropriate combination dictated by the character of the leachate. For a high strength leachate containing high concentrations of both organic and inorganic contaminants, a combination of biological and physical/chemical processes will be needed.

The design of a leachate treatment system will depend not only on the character of the leachate, but on landfill location. For landfills located near a wastewater treatment facility, leachates might be treated by the facility. For landfills distant to a wastewater treatment plant, an aerated lagoon, land application, or complete treatment may be applicable. Each type of system has been successfully demonstrated, either in pilot- or field-scale testing. Regardless of the system selected, leachate recirculation can be beneficial in reducing the organic strength and volume of leachate, as well as reducing treatment costs. Treatment related costs may represent a significant portion of the total landfilling cost, perhaps 25 percent or more.

SECTION 7

ENVIRONMENTAL MONITORING

INTRODUCTION

The Environmental Protection Agency has published a manual titled "Procedures Manual for Ground-Water Monitoring at Solid Waste Disposal Facilities" (EPA/530/SW-611) which provides a guide for supervisory personnel of solid waste regulatory agencies (Fenn, et al., 1977). The Manual establishes the need for ground-water monitoring, assigns priorities for facilities to be monitored and implements and directs a cost-effective, on-going monitoring program. The presented information is offered as guidance and suggested methods only. This section complements the material presented in the manual with additional information which is useful for landfill designers and site operators.

Three topic areas are presented in this section:

1. Monitoring in the vadose zone;
2. Monitoring in the zone of saturation; and
3. Approaches and considerations for monitoring.

The first two sections address available monitoring methods, costs, and the advantages and limitations of the monitoring equipment for these two hydrologic conditions. The final section discusses selection of sampling areas, monitoring frequency, monitoring parameters, and sample collection and preservation technology.

VADOSE ZONE

The vadose zone is that area beneath the top soil and overlying the water table, in which water in pore spaces coexists with air, or in which the geologic materials are unsaturated (Figure 53). Perched water tables may develop above interfaces between layers having greatly different textures. Saturated conditions may also develop beneath recharge sites as a result of prolonged infiltration.

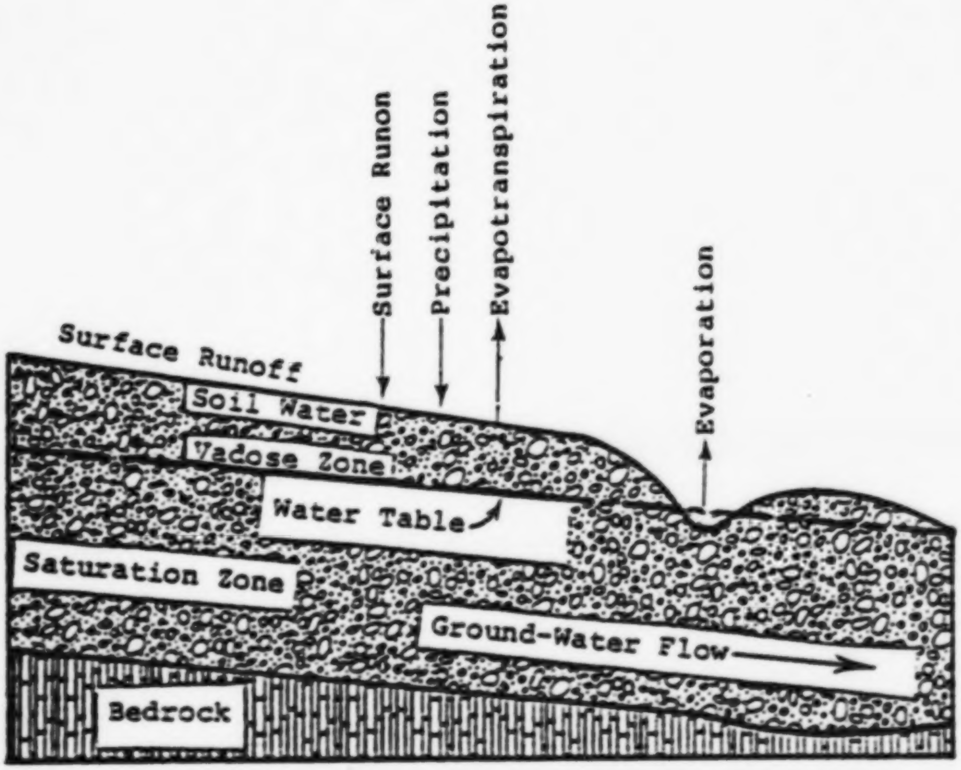
Surface Runon
Precipitation
Evapotranspiration
Evaporation
Surface Runoff
Soil Water
Vadose Zone
Water Table
Saturation Zone
Ground-Water Flow
Bedrock

Figure 53. Hydrogeological cycle.

Wastes applied at the land surface may travel through significant thicknesses of top soil and vadose zone before reaching the water table. Pollutants can be significantly retained or attenuated in the top soil and vadose zone. The storage capacity of the vadose zone for percolating waters may also be great. The vadose zone is an integral component of any leachate monitoring effort; however, in contrast to the large number of studies on water movement in the top soil, parallel studies in the vadose zone have been few.

Monitoring Approaches

Monitoring within the vadose zone may be accomplished through one of several methods: soil sampling, pressure/vacuum lysimeter sampling, and measurement of the volumetric water content of soils in situ. The first two methods provide a means by which water samples may be collected for chemical analyses. Non-sampling methods (i.e., the third type of method) are available for measuring the movement of water through the unsaturated zone. These are especially useful in arid regions to indicate the degree of water saturation and subsequent movement of water in the vadose zone. Non-sampling devices, in conjunction with samplers, can provide an optimum system for sampling when water is flowing through the unsaturated zone. Vadose samplers do not provide this ability to specifically sample during wetting and drying cycles.

Available Monitoring Equipment and Costs

Monitoring equipment for the vadose zone may be classified as sampling or non-sampling. Sampling approaches include pore-water extraction from soil cores and deployment of pressure/vacuum lysimeters to obtain samples of in situ soil moisture. Non-sampling equipment includes tensiometers, psychrometers, moisture blocks and neutron moderation devices.

Water extracted from soil cores may be analyzed for such parameters as major anions, trace metals, total organic carbon (TOC), pH, specific conductance, and other specific constituents. Core samples may provide information on physical characteristics such as soil texture and its change with depth, water content, hydraulic conductivity, and bulk density or water-release curves. These are important if non-sampling approaches are to be used in conjunction with vadose water sampling (Nielsen, et al., 1973).

Soil Sampling--

Equipment and applicability--For shallow sampling of soils and vadose waters, traditional hand augers and bucket type samplers may be used. Hand augers include the Hankenson, Oakfield, and assorted hand piston samplers (Soiltest, Inc., 1976). For deeper sampling, standard drilling equipment and techniques are required (i.e., jetting, water and air rotary, cable tool, augering, and air drilling). Continuous flight augers and air drilling provide the most useable samples. Blake (1969) and Barton (1974) furnish an excellent discussion in their application and use. Methods which utilize drilling fluids preclude obtaining cohesive representative samples for water extraction and complicate correlation of sample quality with depth.

Advantages--The advantages of soil sampling are summarized (Everett, et al., 1976; Fenn, et al., 1977).

- Ease of soil sample collection;
- Accurate vertical and areal sampling locations;
- Chemical species associated with soil solution can be traced downward in a soil profile;
- May be used to measure extent of leachate attenuation at various soil depths;
- Intermittent leachate production permits long interval between sampling periods; and
- Provides for laboratory measurement of soil texture, water content, field capacity, and hydraulic conductivity.

Limitations--Limitations of soil sampling include the following (Everett, et al., 1976; Fenn, et al., 1977):

- Extraction of geochemical fractions from soil samples is a slow, tedious procedure more costly than soil water analysis;
- Extraction procedures are not a proven standardized sampling method for monitoring programs;
- Wetting and drying cycles and changes in redox potential can change chemical reactivity of some soil constituents after collection;
- In situ conditions of soil samples are difficult to maintain; and
- State-of-the-art not documented in leachate studies.

Costs of monitoring--Soil sampling costs vary primarily as a function of sampling depth and sample number. For shallow soil sampling up to 4.5 m (15 ft) in unconsolidated sediments,[*] various hand augers may be used. The cost of hand augers (1981 dollars) updated after Soiltest, Inc. (1976) ranges from $70 to $130. For deeper samples, a hand-driven soil sampler system such as that designed by Veihmeyer (1919) can be used for unconsolidated and consolidated sediments.[†] Basic parts and costs of a Veihmeyer soil sampler are presented in Table 71. A cost diagram for coring in consolidated and unconsolidated formations, sampling every 3 m (10 ft) is given in Figure 54. The cost figures are generalized to cover small-diameter coring of 5 to 7.5 cm (2 to 3 in.) for shallower depths and larger coring sizes of 7.5 to 12.5 cm (3 to 5 in.) for deeper samples, since larger bore diameters are required when drilling to depths greater than 31 m (100 ft). For unconsolidated formations, equipment set-up constitutes a larger part of the sampling costs than for consolidated formation sampling because of more elaborate casing requirements, slumping problems, and drilling fluid differences. At greater depths, the time required for sample procurement overrides the drilling time.

Sampling equipment (e.g., hand augers, rotary rigs, bucket type samplers, sampling tubes) should be thoroughly cleaned prior to reuse to prevent cross-contamination of soil samples. An effective procedure is to wash(scrub) with detergent followed by a tap water rinse. Cost data for cleaning sampling equipment is not available in the literature; however, an approximate assessment can be made for sampling tubes. A conservative estimate is that a technician at $15/hr (including overhead) could clean and rinse ten 1.5-m (10-ft) or 3-m (5-ft) sampling tubes in five hours at a total cost of $75. A comparison of this cost to the replacement cost of ten 1.5-m (10-ft) or 3-m (5-ft) sampling tubes at $721 and $1212, respectively, (Table 71), indicates that service cleaning could be an effective cost/benefit procedure.

Pressure/Vacuum Lysimeters--

Equipment and applicability--Pressure/vacuum lysimeters provide a means for collection of in situ vadose waters. These devices are an alternative to soil sampling for characterization of the water quality of unsaturated soils. These units were developed by Parizek and Lane (1970) and modified by Wood (1973) for sampling depths up to 36 m (118 ft). The design of the Parizek and Lane sampler is shown in Figure 55.

The basic design of the sampling unit consists of a porous ceramic cup capable of holding a vacuum, a rigid-plastic body tube that is equal in length to the desired sampling depth, and two sampling tubes leading to the surface. When placed in the soil, the pores in the ceramic cup become an extension of the pore space of the soil so that the soil-water content in the soil and the cup become equilibrated at the existing soil-water pressure. After emplacement, a vacuum is applied to the cup. Soil solution is drawn into the sampler under this gradient. The vacuum is then released and pressure is applied, forcing the accumulated water sample to the surface through the outlet sampling tube. Although pressure/vacuum samplers have limitations, they are considered to be the best available device for sampling within the vadose zone.

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[*] unconsolidated sediments: sand and gravel

[†] consolidated sediments: limestone, sandstone, and dolomite

TABLE 71. BASIC PARTS AND COSTS OF A VEIHMEYER SOIL SAMPLER (Veihmeyer, 1919)

Part[*]	Cost[†]
Tube, 1.5 m (5 ft)	$72.10
Tube, 3 m (10 ft)	121.20
Tip, Type A, general use	36.90
Drive head	41.50
Drop hammer, 6.8 kg (15 lb)	102.75
Puller jack and grip[‡]	231.50
Total	$605.95

[*] One of each part is needed. Manufactured by Hansen Machine Works, 334 N. 12th Street, Sacramento, CA 95815.

[†] Updated to December, 1981.

[‡] Recommended primarily for deep soil sampling.

Advantages--The advantages of pressure/vacuum lysimeters are as follows (Fenn, et al, 1977; Everett, 1980; Wilson, 1980):

- Best available techniques for obtaining samples of the soil solution in situ;
- Inexpensive sampling device of great reliability;
- Inexpensive installation; and
- Samples can be collected at a central point.

Limitations--Limitations of pressure/vacuum lysimeters include the following (Hansen and Harris, 1975; Fenn, et al., 1977;

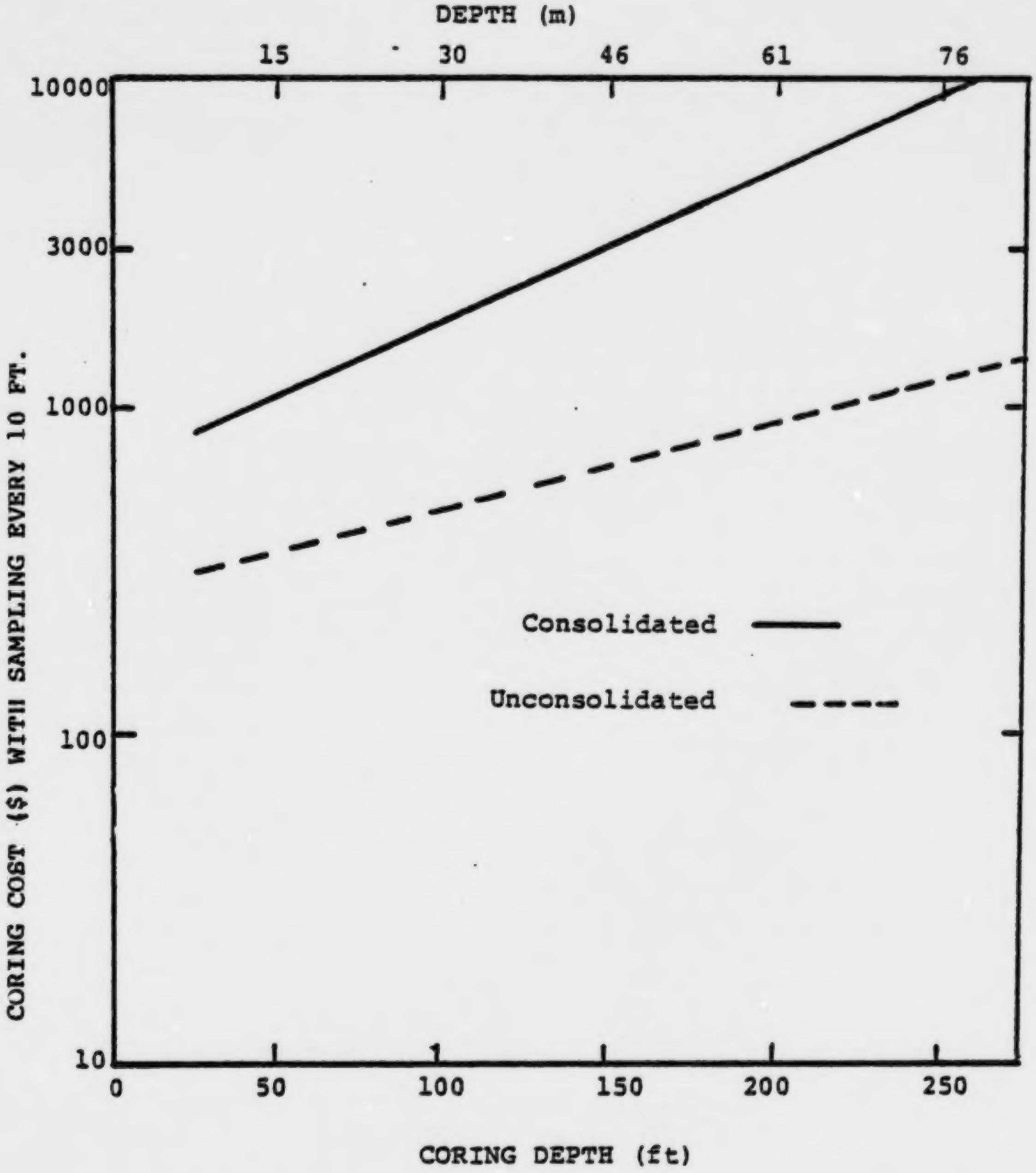

Figure 54. Generalized cost of coring in unconsolidated and consolidated formations, December, 1981 (after Everett, et al., 1976).

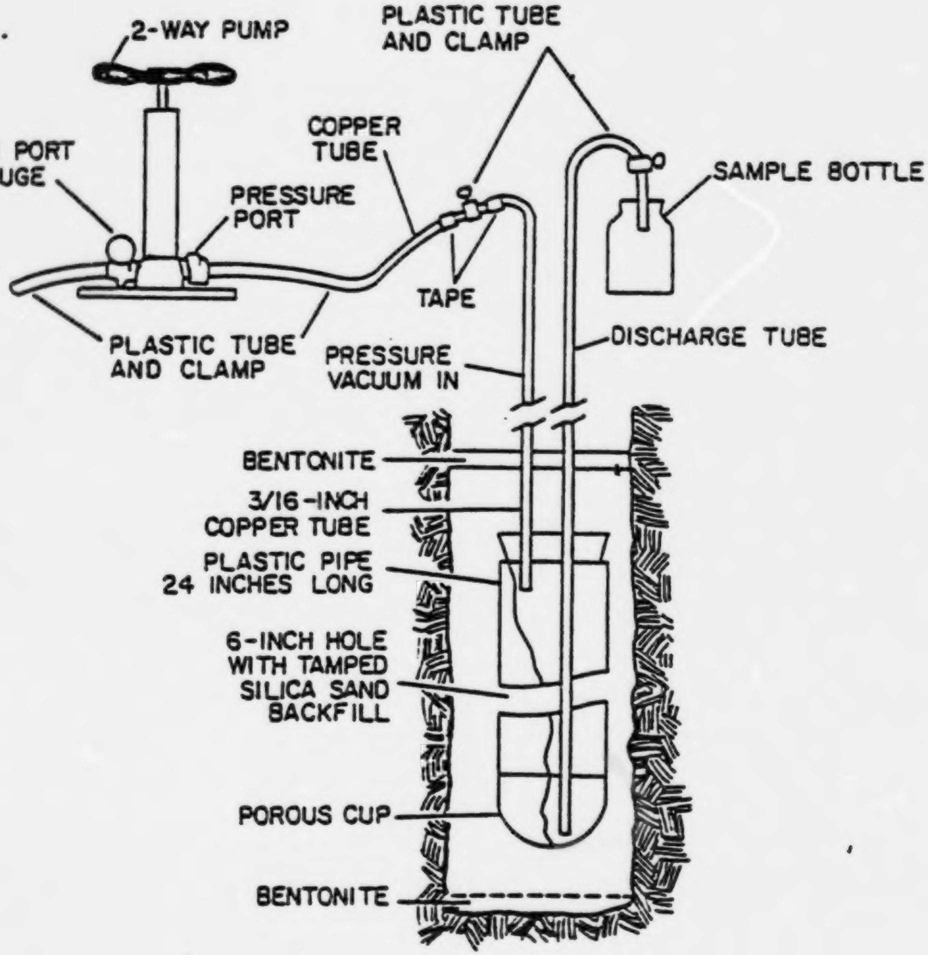
2-WAY PUMP
PLASTIC TUBE AND CLAMP
COPPER TUBE
VACUUM PORT AND GAUGE
PRESSURE PORT
SAMPLE BOTTLE
TAPE
PLASTIC TUBE AND CLAMP
PRESSURE VACUUM IN
DISCHARGE TUBE
BENTONITE
3/16-INCH COPPER TUBE
PLASTIC PIPE 24 INCHES LONG
6-INCH HOLE WITH TAMPED SILICA SAND BACKFILL
POROUS CUP
BENTONITE

Figure 55. Pressure/vacuum lysimeter (after Parizek and Lane, 1970).

Wilson, 1980):

- Suction cups are capable of sampling only at pressure greater than about -1.0 atmosphere; therefore, samples cannot be obtained over the entire range of soil-water pressures;
- Chemical transformations within the ceramic cup are possible;
- Small quantities of samples are collected over long periods of time preventing short-term analysis of concentrations as a function of time;
- Macromolecules may be screened out altering the sample and plugging of the ceramic cup;
- Repairs are difficult in situ, and
- Samples represent only a small area; a localized region of pollutant leakage may occur undetected.

Costs of monitoring--Commercially available lysimeters may have one of a variety of porous cups and plastic tube designs. (Soiltest, Inc., 1976; Soil Moisture Equipment Corp., 1978; Timco Manufacturing Company, Inc., 1980) Depending upon the length of the ceramic tip and plastic tube, individual units cost $50 to $60 (1981 dollars) based on 1978 costs (Soil Moisture Equipment Company, 1978). Plastic withdrawal and air injection tubing costs 15¢ to 20¢ per ft. Bentonite seals and silica sand packing costs $2.00 to $3.00 per installed unit. Figure 56 gives the capital costs of pressure/vacuum lysimeters for various depths and sample space densities. The costs presented in Figure 56 do not include installation.

Tensiometers--

Equipment and applicability--Tensiometers are used to measure soil-water pressures during unsaturated flow conditions; soil-water pressures may be employed to estimate soil-water content through the development of suitable soil-water characteristic curves (Nielsen, et al., 1973). Tensiometers consist of a porous ceramic cup connected to a rigid plastic tube which in turn is connected to a manometer with small diameter tubing. The internal volume of the system is filled with water except for the mercury reservoir (Figure 57). When properly emplaced in the soil, the pores in the ceramic cup form a continuum with the pores in the soil. Water moves either in or out of the tensiometer system until an equilibrium is attained across the ceramic cup. Changes in pressure are measured by the mercury manometer. Holmes, et al., (1967) described their use and operation. Watson (1967) presented a design for a tensiometer-

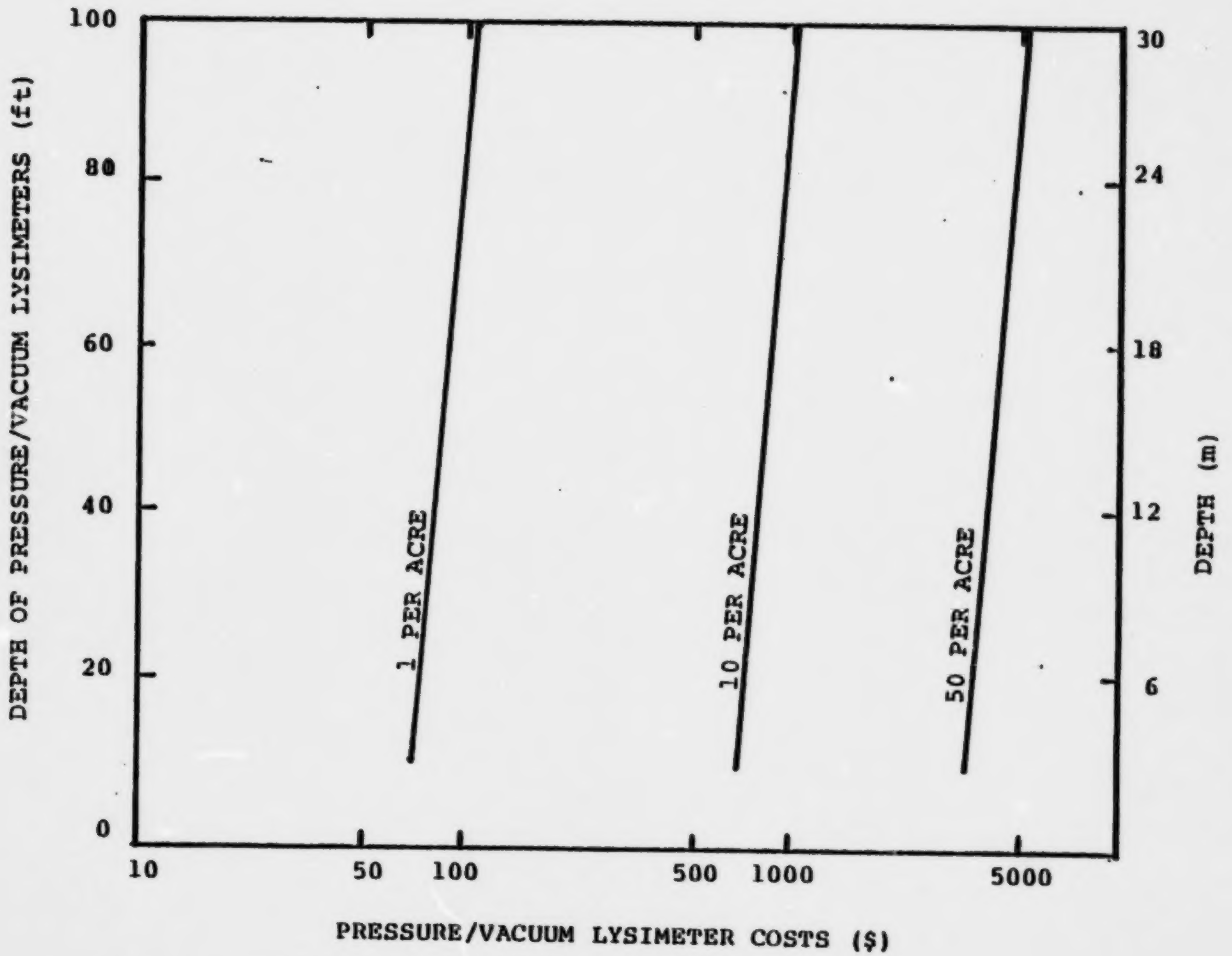

Figure 56. Cost of pressure/vacuum lysimeters for various depths and sampling densities, December, 1981 (after Everett, *et al.*, 1976).

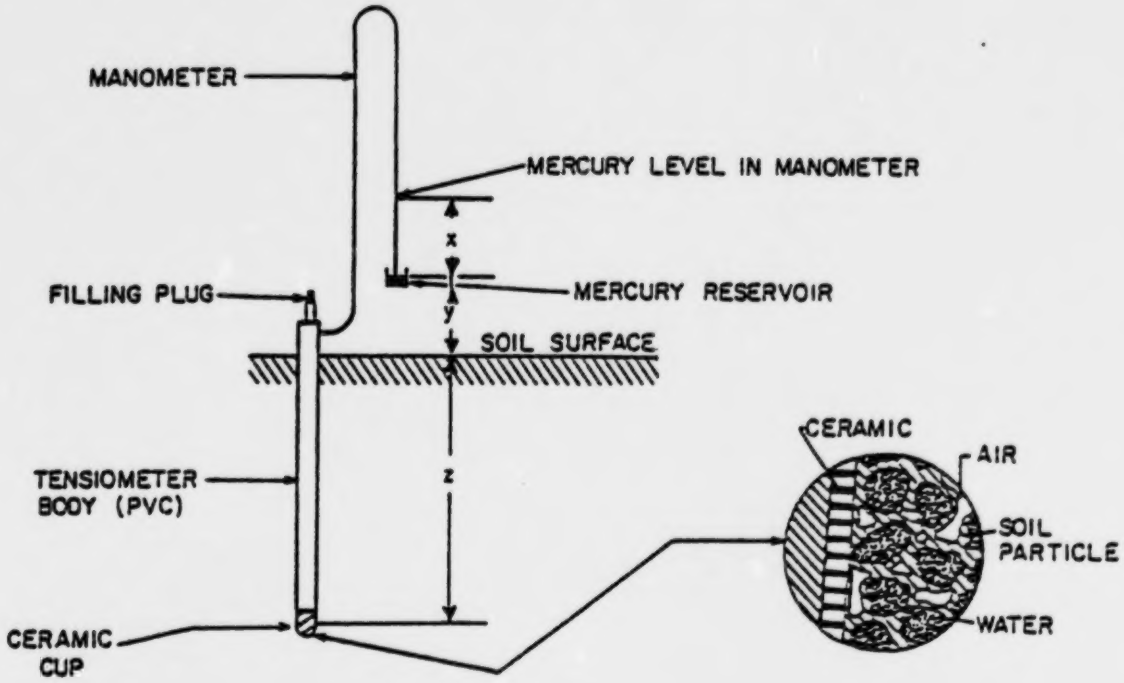
MANOMETER
MERCURY LEVEL IN MANOMETER
x
FILLING PLUG
MERCURY RESERVOIR
y
SOIL SURFACE
z
TENSIOMETER BODY (PVC)
CERAMIC CUP
CERAMIC
AIR
SOIL PARTICLE
WATER

Figure 57. Schematic representation of tensiometer and section through the ceramic cup (after Nielson, et al., 1973).

pressure transducer system, in lieu of the mercury system, to permit the conversion of soil water pressure into any equivalent electrical resistance. Figure 58 illustrates this system.

Advantages--The advantages of tensiometers include the following (Gairon and Hadas, 1973; Oaksford, 1978; Brakensiek, et al., 1979; Wilson, 1980):

- Provide continuous in place measurements of water content;
- Successive measurements are obtained;
- Inexpensive and simple; and
- Transducer units respond fairly rapidly to water content changes.

Limitations--Limitations of tensiometers include the following (Holmes, et al., 1967; Schmugge, et al., 1980; Wilson, 1980; Everett, 1981);

- Units fail at the air entry value of the ceramic cup, generally about 0.8 atmospheres;

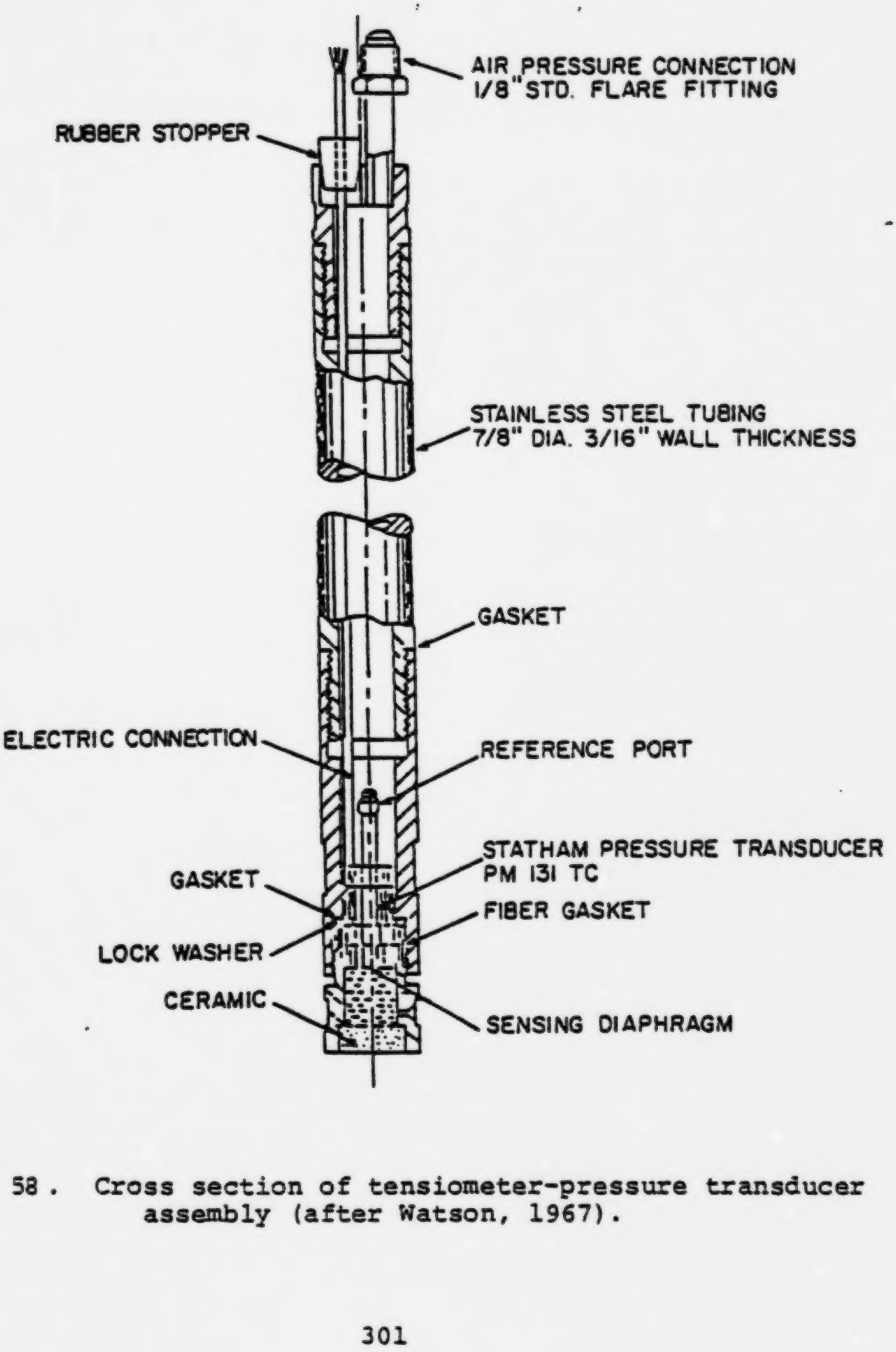
AIR PRESSURE CONNECTION
1/8" STD. FLARE FITTING
RUBBER STOPPER
STAINLESS STEEL TUBING
7/8" DIA. 3/16" WALL THICKNESS
GASKET
ELECTRIC CONNECTION
REFERENCE PORT
STATHAM PRESSURE TRANSDUCER
PM 131 TC
GASKET
FIBER GASKET
LOCK WASHER
CERAMIC
SENSING DIAPHRAGM

Figure 58. Cross section of tensiometer-pressure transducer assembly (after Watson, 1967).

- Results are subject to hysterisis, i.e., different results are obtained for drying and wetting cycles;

- If proper contact is not made between cup and media, units will not operate properly;

- Sensitive to temperature changes; and

- Difficult to install at great depth in vadose zone.

Costs of monitoring--The cost of a tensiometer utilizing a mercury manometer including a 1.8-m (6-ft) rigid PVC tube is approximately $54 (1981 dollars) based on 1974 costs (Everett, et al., 1976). Figure 59 gives the capital costs of tensiometers for various depths and sample space densities. The costs presented do not include installation.

Psychrometers--

Equipment and applicability--Tensiometers cannot be used to measure soil-water pressure below about -1.0 atmosphere because of air-entry problems. In recent years, progress has been made in developing the thermocouple psychrometer for this purpose. According to Watson (1974), in situ pressure measurements down to -30 atmospheres are possible with these units. The principle of psychrometric measurement of soil-water potential as discussed by Rawlens and Dalton (1967), is that a relationship exists between soil-water potential and the relative humidity of soil water.

Psychrometers employ a porous bulb with a chamber to measure the relative humidity of the soil, a sensitive thermocouple, a heat sink, a reference electrode, and associated electronic circuitry. Such units use the principle of Peltier cooling to lower the temperature of one junction of the thermocouple below the dewpoint, thereby allowing an evaluation to be made of the relative humidity (Wilson, 1980). Two types of thermocouple psychrometers are available. One type, which is installed in access tubes, is constructed by mounting psychrometers in porous cups at the base of tubing. This unit may be withdrawn for recalibration. The second type, shown in Figure 60 , is called a "sealed-cup" psychrometer by Merrill and Rawlins (1972).

Advantages--The advantages of psychrometers are as follows (Rawlins and Dalton, 1967; Merrill and Rawlins, 1972; Enfield, et al., 1973; Schmugge, et al., 1980):

- In situ pressure measurements are possible down to -30 atmospheres, permitting the determination of water contents in the very dry range;

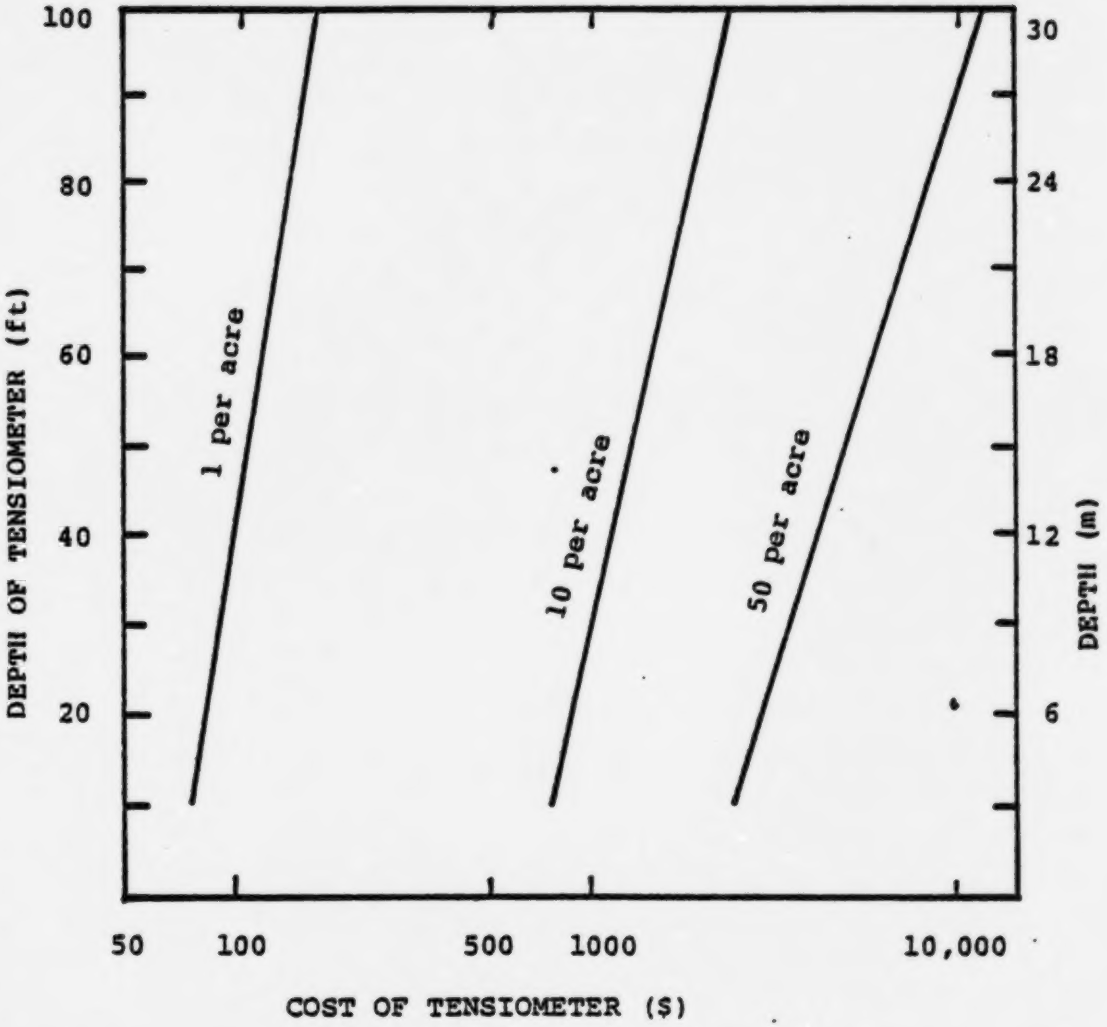

Figure 59. Cost of tensiometer and electrical meter for various depths and sampling densities, December, 1981 (after Everett, et al., 1976).

- Permits continuous recording of pressures and water contents;
- Can be interfaced with portable or remote data collection systems; and
- Some units have been installed to great depth (down to 91 m (300 ft).

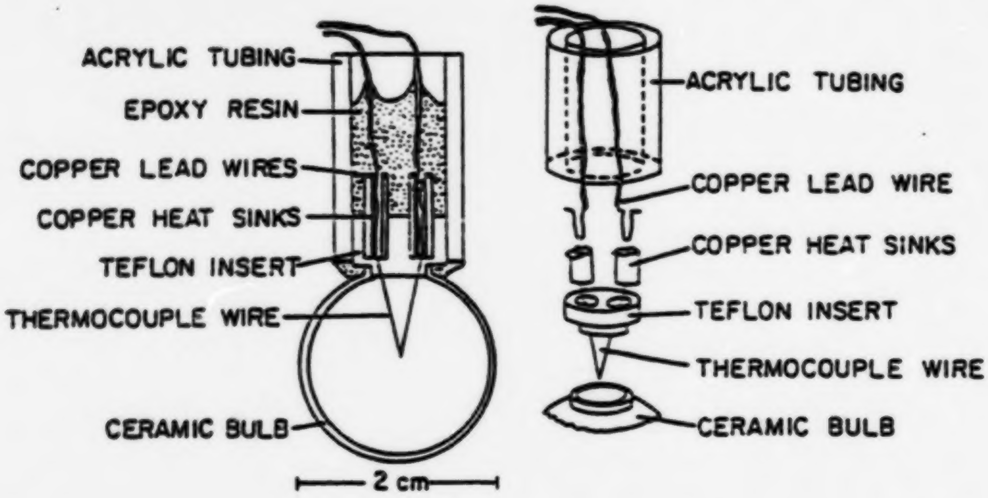
ACRYLIC TUBING
EPOXY RESIN
COPPER LEAD WIRES
COPPER HEAT SINKS
TEFLON INSERT
THERMOCOUPLE WIRE
CERAMIC BULB
2 cm
ACRYLIC TUBING
COPPER LEAD WIRE
COPPER HEAT SINKS
TEFLON INSERT
THERMOCOUPLE WIRE
CERAMIC BULB

Figure 60.. Soil psychrometer (after Merrill and Rawlins, 1972).

Limitations--Limitations of psychrometers include the following (Campbell, et al., 1973; Hanks and Ashcroft, 1980; Schmugge, et al., 1980; Wilson, 1980):

- Results are subject to hysteresis;
- Good contact between bulk and surrounding media may be difficult to obtain;
- Provide point measurements only;
- May be difficult to obtain accurate calibration curves for deep regions of the vadose zone; and
- Fragile, requires great care in installation.

Costs of monitoring--The cost of a psychrometer is approximately $80 (1981 dollars) based on 1976 costs (Soiltest, Inc., 1976). Figure 61 shows the capital costs of psychrometers including access tubes for various depths and sample space densities. The costs presented do not include installation.

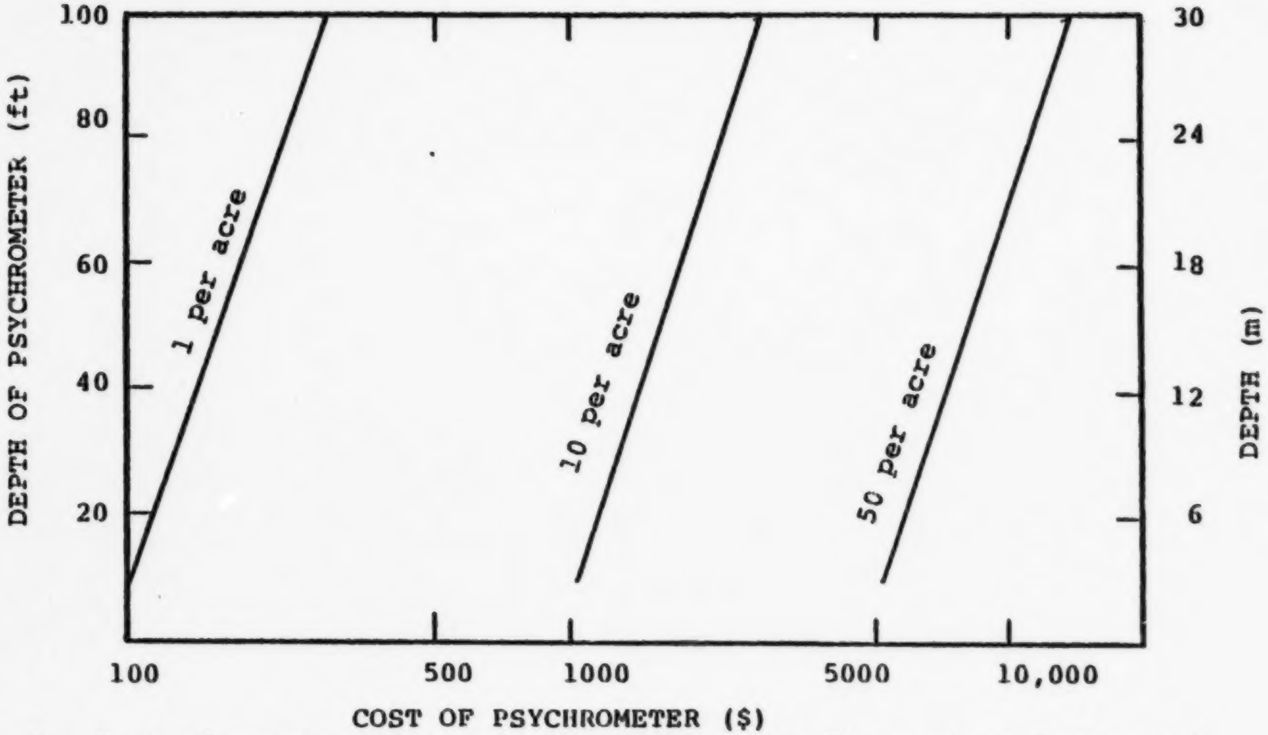

Figure 61. Cost of psychrometer and electrical meter for various depths and sampling densities, December, 1981 (after Soiltest, Inc., 1976).

Electrical Resistance Blocks--

Equipment and applicability--Electrical resistance blocks (moisture blocks) consist of electrodes embedded in a suitable porous material such as gypsum, fiberglass, or nylon cloth. Soil moisture contacts are connected by rubber-covered leads to the surface. A schematic representation of a gypsum moisture block is shown in Figure 62. Resistance blocks are used to measure either soil-water content or soil-water pressure. Water content (or negative pressure) within the blocks responds to the water content (or suction) of the soil with which the blocks are in intimate content and the electrical resistance properties of the blocks change correspondingly. Moisture blocks calibration involves evaluating resistance readings against a range of soil-water contents or negative pressures. Moisture blocks may be used to measure soil-water pressure if suitable soil-water characteristic curves are available (Everett, 1981).

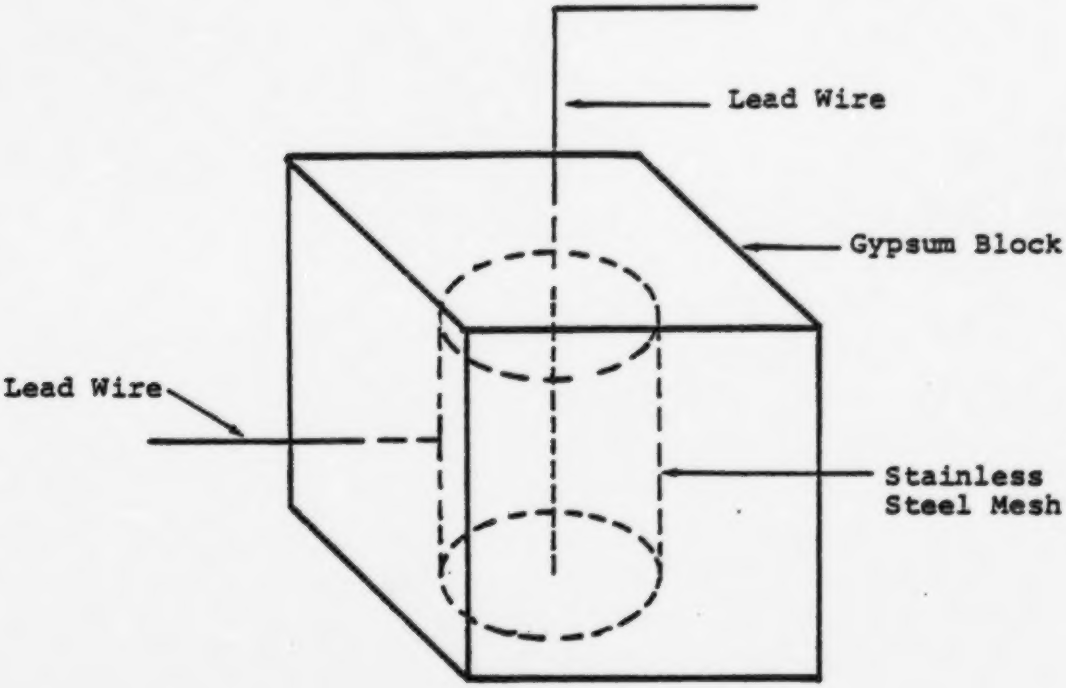

Figure 62. Gypsum electrical resistance block

Wigh (1971) studied the use of a MC-300A soil moisture meter and block at a sanitary landfill and found that changes in the moisture content of the refuse could be determined. Use of such data as part of a monitoring strategy has not been widely utilized, although it appears promising. Schmugge, et al., 1980 suggested similar networks as an early warning system to trigger implementation of a synoptic vadose sampling program.

Advantages--The advantages of electrical resistance blocks are as follows (Holmes, et al., 1967; Brakensiek, et al., 1979; Schmugge, et al., 1980; Everett, 1981):

- Inexpensive and simple;
- Can be calibrated for either suction or water content;
- Can be used at soil water pressures less than -0.8 atmospheres;
- Can be interfaced with portable data collection systems; and
- Precision is good.

Limitations--Limitations of electrical resistance blocks include the following (Phene, et al., 1971; Gairon and Hadas, 1973; Wilson, 1980; Everett, 1981):

- May be difficult to install at great depth in vadose zone and maintain good contact;
- Requires calibration for each textural type in profile;
- Lack of sensitivity in wet range;
- Sensitivity to soil salinity;
- Gypsum blocks deteriorate badly in certain media;
- Calibration curves of some units shift with time; and
- Time lag in response.

Costs of monitoring--Moisture blocks provide an inexpensive means of developing a grid to determine the moisture content of a soil. Because of their low cost, a dense array of moisture blocks may be placed under a landfill to detect leakage. The cost of an electrical resistance block is approximately 8 dollars (1981 dollars). The cost of a soil moisture meter is approximately $250 based on 1981 costs (Everett, et al., 1976). The capital costs, excluding emplacement, of electrical resistance blocks, leads, and a soil moisture meter for various depths and sample space densities are given in Figure 63.

Neutron Moisture Logging--

Equipment and applicability--Neutron moisture logging is employed to measure changes in the volumetric water content within a soil horizon. It is also used to delineate perched water zones and to estimate flow rates.

The neutron thermalization method is based on the principle that high energy neutrons are slowed down (thermalized) in their trajectories primarily by collisions with light atomic nuceli, especially those in hydrogen atoms. Instrumentation used to measure water content by neutron thermalization requires three principal components:

(1) A source of fast neutrons;

(2) A detector of slow neutrons; and

(3) An instrument to determine the count rate from the detection equipment (Wilson, 1981).

The basic components of a neutron moisture logger are shown in Figure 64.

In operation, source of high energy neutrons (e.g., amerecium-beryllium) in a down-hole tool is lowered into an access well. Water in the vadose zone slows down the fast neutrons which are captured by a detector in the tool. Counts measured by a ratemeter are converted into volumetric water content by an appropriate calibration relationship. Successive readings show temporal changes in water storage at different depths (Everett, 1981; Wilson, 1981).

The use of the neutron moisture logger normally requires a cased well. The inside diameter of wells should be as close as possible to the outside diameter of the probe. During installation, it is essential to establish a tight fit between the borehole wall and the casing, to minimize water migration via the annulus. Drilling mud should be used since it interfers with water content observations. In situations requiring a tight fit, drilling techniques which do not require a drilling mud (jetting or augering) are used (Everett, 1981).

Advantages--The advantages of neutron moisture logging are as follows (Holmes, et al., 1967; Brakensiek, et al., 1979; McGowan and Williams, 1980; Wilson, 1980; Everett, et al., 1981):

- Rapid;

- Water content profiles can be obtained in situ;

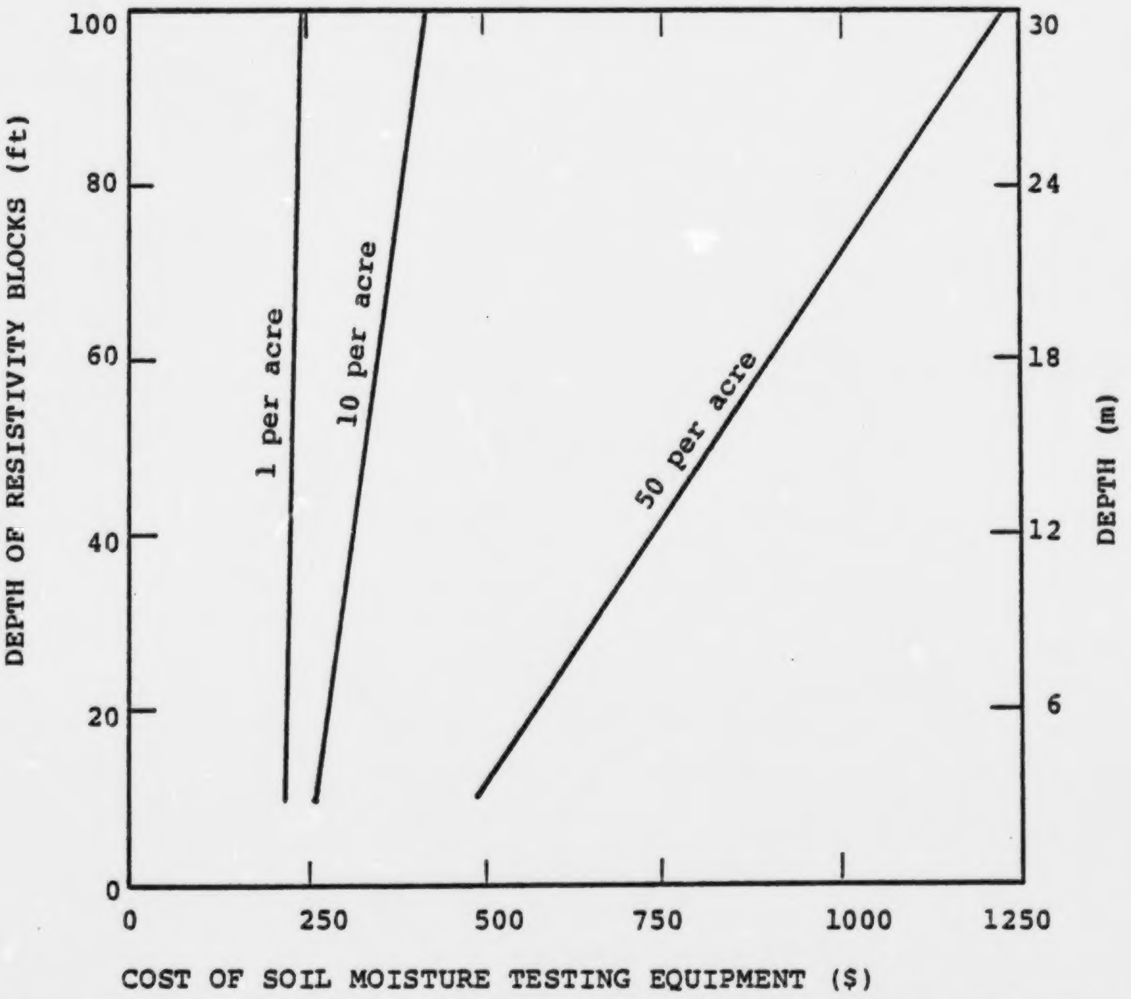

Figure 63. Cost of multiple electrical resistance blocks and soil moisture meter for various depths and sampling densities, December 1981 (after Everett, et al., 1976).

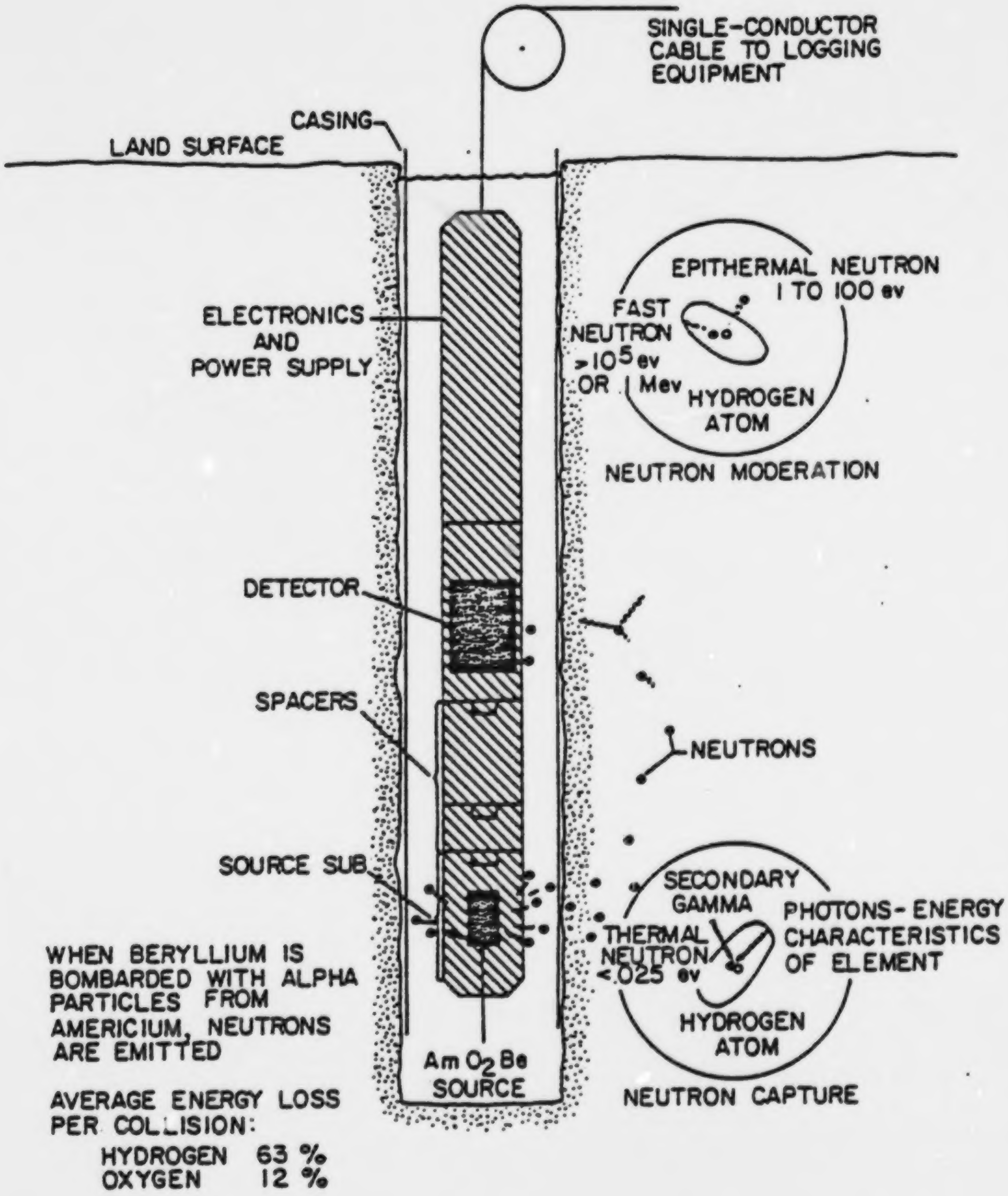
SINGLE-CONDUCTOR CABLE TO LOGGING EQUIPMENT
CASING
LAND SURFACE
ELECTRONICS AND POWER SUPPLY
EPITHERMAL NEUTRON 1 TO 100 ev
FAST NEUTRON >10⁵ ev OR 1 Mev
HYDROGEN ATOM
NEUTRON MODERATION
DETECTOR
SPACERS
NEUTRONS
SOURCE SUB
SECONDARY GAMMA
PHOTONS-ENERGY CHARACTERISTICS OF ELEMENT
THERMAL NEUTRON <.025 ev
HYDROGEN ATOM
NEUTRON CAPTURE
Am O2 Be SOURCE
WHEN BERYLLIUM IS BOMBARDED WITH ALPHA PARTICLES FROM AMERICIUM, NEUTRONS ARE EMITTED
AVERAGE ENERGY LOSS PER COLLISION:
HYDROGEN 63 %
OXYGEN 12 %

Figure 64. Equipment and principles of neutron moisture logging (after Keys and MacCary, 1971).

- Can be used to locate perched ground-water zones together with their growth and dissipation;
- Water content changes at a given depth in a succession of wells may provide data on lateral flow velocities; and
- Can be interfaced with portable data collection systems.

Limitations--Limitation of neutron moisture logging include the following (Brakensick, *et al.*, 1979; Schmugge, *et al.*, 1980; Wilson, 1980; Everett, 1981):

- Expensive, requiring the purchase or leasing of equipment;
- Cannot relate results exactly to a specific depth;
- Fast neutrons are moderated by other constituents besides hydrogen in water, e.g., chloride and boron. Accuracy may be affected;
- An indirect method requiring a difficult calibration procedure; and
- Cannot be used to infer water movement in regions where storage changes do not occur.

Costs of monitoring--The cost of a neutron moisture logger including the neutron source, slow neutron detector, and surface counter is about $5,500 (1981 dollars), updated after Everett, 1976. Figure 65 gives the cost of neutron moisture logging with various numbers of well loggings performed at depth intervals of 2 feet. The costs in Figure 65 include only the time required to conduct the log; they do not include capital costs for the equipment and installation.

ZONE OF SATURATION

The zone of saturation is that portion of the ground-water system in which available pore space is occupied by water (Figure 53). The zone of saturation may include both permeable and impermeable earth layers.

Monitoring Approaches

Monitoring within the zone of saturation has traditionally been accomplished with screened wells. A serious limitation of this method is its inability to provide a representative vertical cross section of contamination within the aquifer due to vertical mixing within the well casing. More recently developed equipment

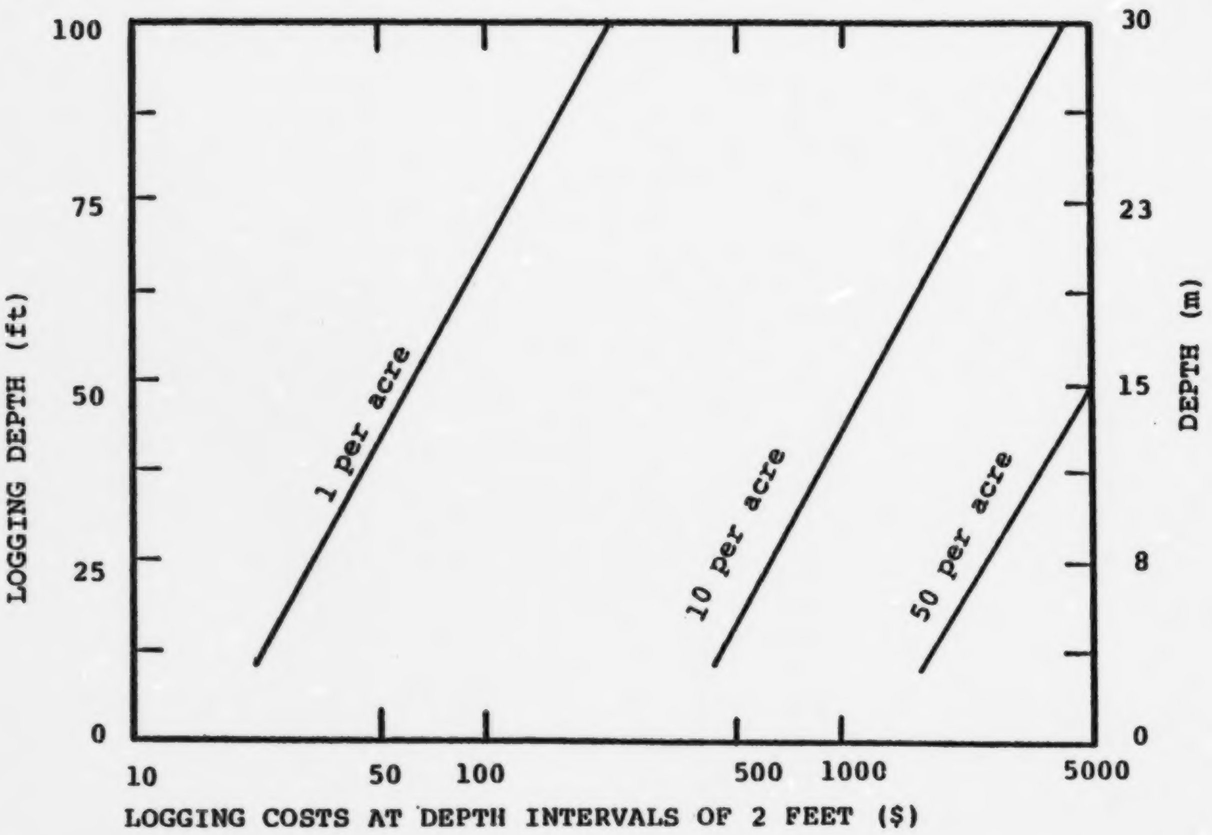

Figure 65. Cost of neutron logging for various depths and sample space densities, December, 1981 (after Everett, et al., 1976).

allows for multiple sampling points throughout the aquifer.

Irrespective of the selection of equipment and the location of samplers, accurate hydrogeologic information is required. At a minimum, the hydraulic gradient must be known. The hydraulic gradient is obtained by measurement of not less than 3 wells which form a triangle on the perimeter of the site. Water level measurements can then be used to determine approximate flow patterns. A more complete hydrologic understanding of the site will usually be obtained with additional wells and water level measurements. Figure 66 was developed to illustrate the use of this method.

Determination of ground-water flow patterns represents the minimum data required for establishing a monitoring program. Other critical data are the characteristics of the geological formation, water level fluctuations, and soil types. Various researchers have provided detailed procedures describing the information required and the approaches by which the necessary physiographic data for a disposal site may be obtained (Todd, et al., 1976; Sendleim and Yazicigil, 1981). The success of the monitoring program is usually closely related to the results of these pre-monitoring site characterizations.

Available Monitoring Equipment and Costs

For purposes of discussion, monitoring equipment within the zone of saturation is described in three categories:

1. Wells with the capacity to sample at a single depth;
2. Multi-sampling wells for sampling at different depths; and
3. Piezometers which are designed to obtain samples utilizing air lift methods.

Well Construction--

Wells are constructed by the following principal methods (Everett, et al., 1976):

- Driven;
- Augered;
- Jetted;
- Cable tool (percussion); and
- Rotary drilled (i.e., direct-circulation, reverse-circulation, air-circulation).

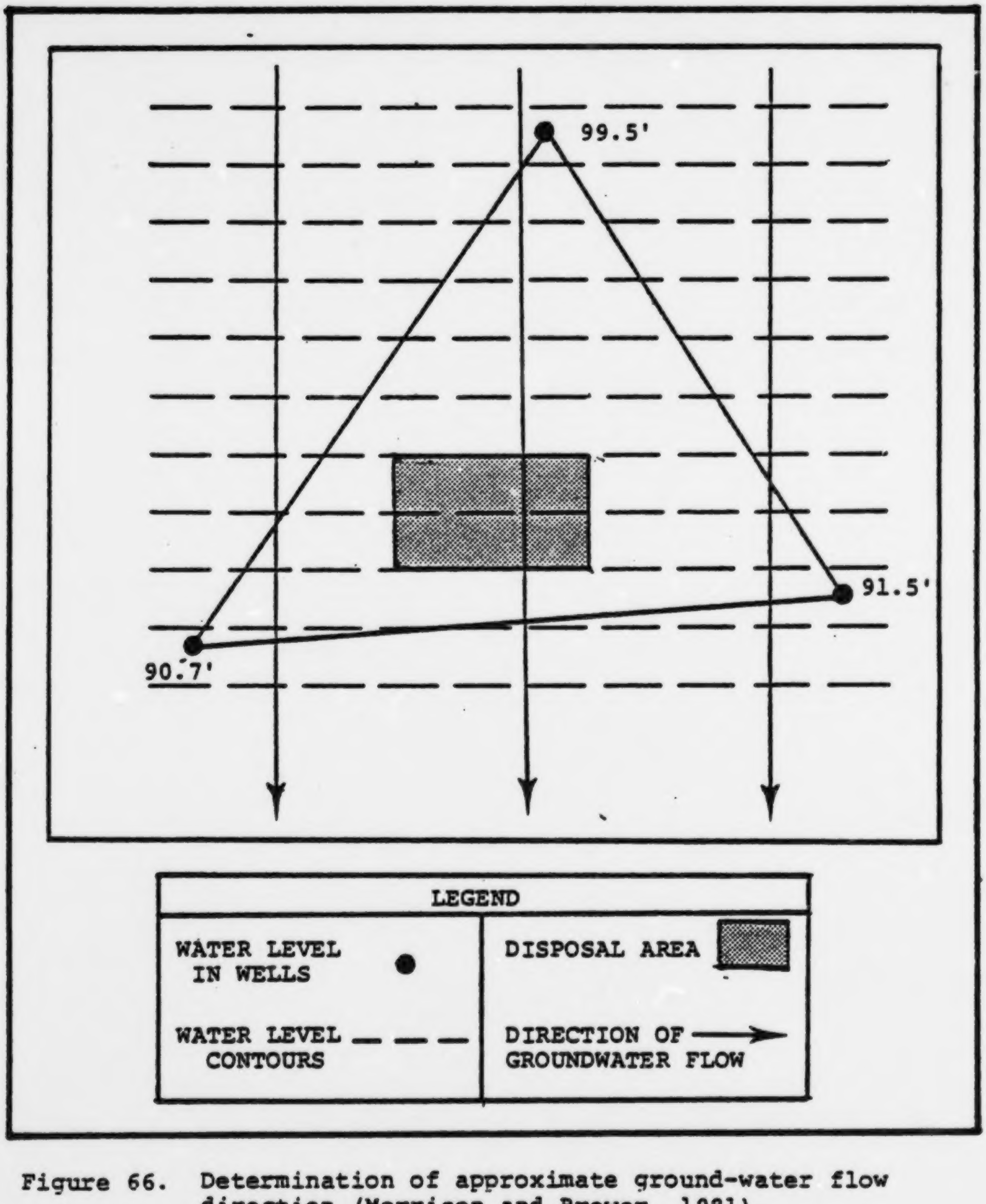
99.5'
91.5'
90.7'
LEGEND
WATER LEVEL IN WELLS
DISPOSAL AREA
WATER LEVEL CONTOURS
DIRECTION OF GROUNDWATER FLOW

Figure 66. Determination of approximate ground-water flow direction (Morrison and Brewer, 1981).

Driven--Driven wells (i.e., sandpoint wells) consist of a casing 3 to 5 cm (1½ to 2 in.) in diameter attached to a steel drive point which is slotted. These wells are driven to depths generally up to 9 m (30 ft). Wells points are most useful in unconsolidated soils devoid of rocks or pebbles or deposits of hard materials. The chief disadvantage of driven wells are that construction is slow and difficult when tightly compacted materials are encountered; however, they are useful for monitoring shallow aquifers in unconsolidated rock.

Augered--Shallow wells may be augered by hand or power equipment. Hand augers are useful in penetrating unconsolidated deposits less than 4.5 m (15 ft). Monitoring wells to depths of 46 to 61 m (150 to 200 ft) with holes diameters ranging from 5 to 81 cm (2 to 32 in.) can be constructed in unconsolidated formations by use of a drilling rig equipped with a power auger. Casing is usually required for deeper holes and during drilling into the saturated zone. Where hollow-stem augers are used, a casing and well point can be set inside the hollow stem to the required depth.

Jetted--Water jetting and associated hydraulic approaches use a high-velocity stream of water or jetting fluid for drilling. The water stream loosens the material and transports it to the surface. The jetting method is most successful in sandy soils with shallow water tables. The technique is ineffective against hard rock and boulders; compacted clay and hardpan also presents problems. The casing is usually sunk as the jetting proceeds, and if too much resistance is encountered, the casing may be driven. Water jetted wells are usually of small diameter and less than 46 m (150 ft) deep.

Cable tool (percussion)--In cable tool (percussion) drilling, the hole is formed by the percussion and cutting action of a drilling kit that is alternatively raised and dropped. The drill cuttings are removed at intervals by bailer or sand pump. Casing is not required in consolidated rock, but in unconsolidated formations, casing is driven down the hole during drilling. The bottom of the casing is fitted with a hardened steel drive shoe. As the casing is driven in unconsolidated formations, frictional forces increase until the casing cannot be driven further without the use of special tools. Small cable tool rigs can usually drill a 13-cm (5-in.) diameter hole to a depth of about 91 m (300 ft).

Rotary drilling-- Rotary drilling has virtually no depth limit for most ground-water systems. In direct-circulation (conventional) rotary drilling, the drill stem rotates downward as drilling mud is pumped through the inside of the drill pipe and out through the openings in the kit. Biodegradable muds are presently in use which do not contaminate the waterbearing formation. The principal advantages of direct-rotary drilling are speed, great depth capability, and the ability to run electric logs in an open hole prior to casing installation.

In the reverse-circulation rotary method, the drilling fluid is introduced down the hole outside of the drill pipe. The drill cuttings are pumped up through the drill pipe as drilling proceeds and are separated from the drilling fluid at the surface.

The air-circulation rotary method is similar to the other rotary methods, with the exception that compressed air is used instead of drilling mud to bring the cuttings to the surface. In consolidated rock, the conventional drill kit can be replaced with a pneumatic drill kit to speed up the drilling.

Costs of Drilling--

In a well-constructed program, the cost of drilling can be separated from well development and can be approximated. In general, drilling costs range from \$75 to \$200/hr or \$16 to \$66/m (\$5 to \$20/ft)*. The cost of drilling will vary primarily with hole diameter, type of geologic formation and depth of the well. Drilling costs for 12- to 61-m (40- to 200- ft) depths in unconsolidated formations for various well diameters are given in Figure 67. Similarly, drilling costs for consolidated formations are given in Figure 63.

Well Development--

Well casing--Campbell and Lehr (1973) summarized five types of casing material used for monitoring wells:

1. Standard pipe;
2. Line pipe;
3. Reamed and drifted pipe;
4. Drive pipe; and
5. Water-well casing.

Monitoring well economics and possible interference with identified pollutants will usually determine material selection. For purposes of water sampling, plastic (e.g., PVC, fiberglass-reinforced plastic pipe) well screen and casings are preferred because of their chemical inertness which minimizes contamination of collected water samples.

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* Recent survey (1981) by Calscience Research, Inc., in Southern California area.

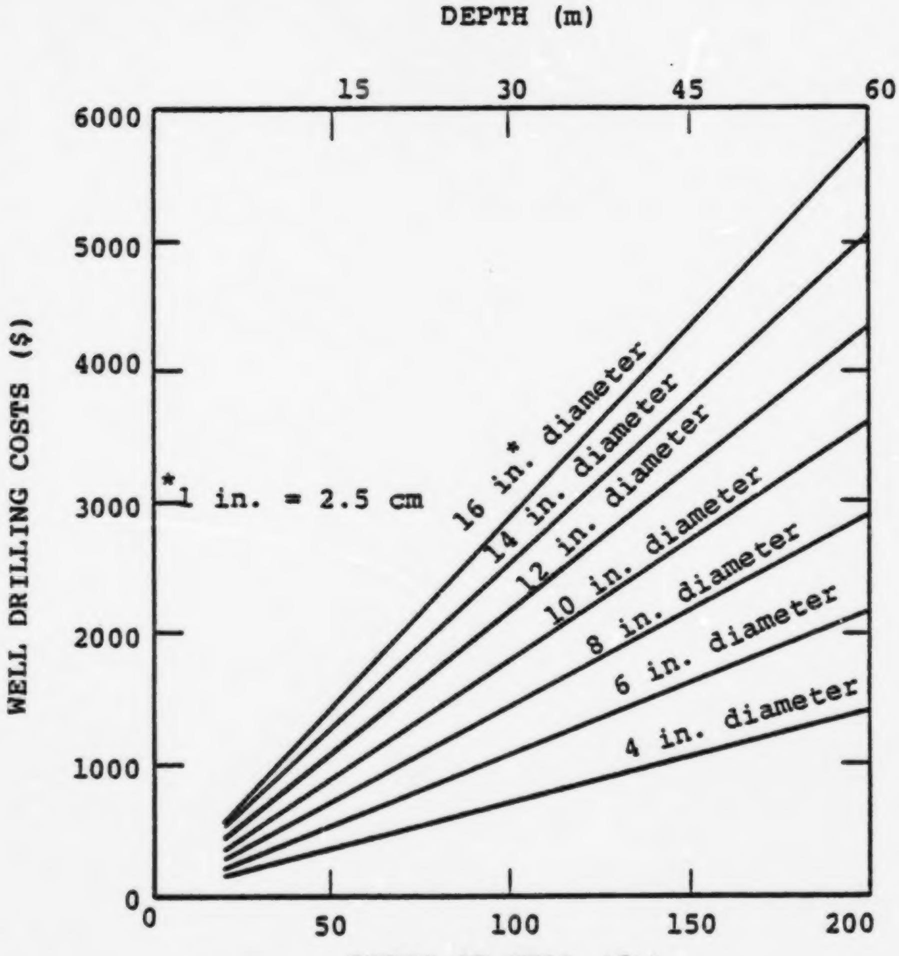


Figure 67. Costs of well drilling in unconsolidated formations, December, 1981 (after Everett, et al., 1976).

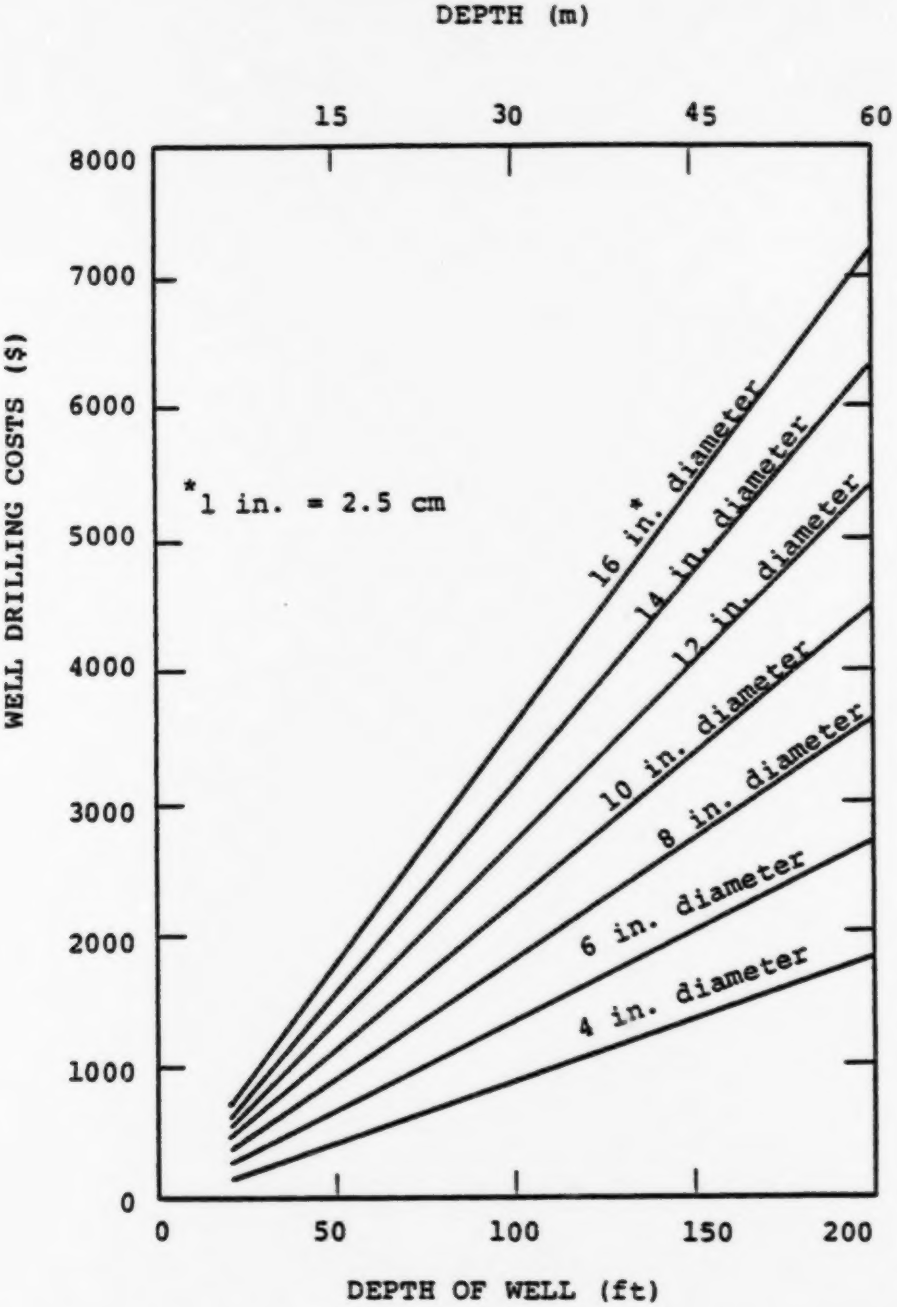


Figure 68. Costs of well drilling in consolidated formations, December, 1981 (after Everett, et al., 1976).

Other well casing materials are ceramic tile, concrete, and asbestos cement. Less common materials such as stainless steel, cupro-nickel alloys, silicon bronze, aluminum, and other non-ferrous metals, can be used for casing in special situations when the natural soil and water quality conditions dictate their employment (EPA, 1975).

Polyvinyl chloride (PVC) casing has increased in use in recent years. Maximum installation depths for PVC casing are normally less than 61 m (200 ft). PVC casing costs for diameters of 5 to 25 cm (2 to 10 in.) and lengths of 6 to 61 m (20 to 200 ft) are given in Figure 69.

Well screens--In unconsolidated materials, openings in the casing must be provided to permit entrance of water. The size of perforations, slots, or screen openings are chosen with respect to particle size distribution of the water bearing-zones. Entrance velocity into the screen is usually recommended to be less than 0.1 to 0.2 feet per second. Well screens are available in iron, stainless steel, brass, fiberglass, and PVC (Everett, 1976; EPA, 1975).

Gravel packing-- After casing and screen have been installed, a gravel pack is placed around the screen. The object of gravel packing is to provide a zone of material coarser than the natural material. The gravel pack in conjunction with appropriate-sized screen openings, retain fine material while permitting water to enter without excessive head loss (EPA, 1975).

Gravel packing is also used for formation stabilization. Since drilling by the rotary method through unconsolidated materials results in a hole somewhat larger than the outside diameter of the casing, the annular space around the well screen is gravel packed to prevent silt and clay above the water table from caving or slumping into the well producing zone (Everett, et al., 1976).

Well sealing (grouting)--Well sealing or grouting isolates the screened zone. Grouting materials include portland cement, bentonite, pazzolana, perlite, diatomaceous earth, Gilsonite, and mixtures of these. From a sampling standpoint, well sealing is vital to protect the producing zone from contamination. Without the seal, rainwater would infiltrate the backfill which could result in dilution of the collected samples, or fluids could move downward, from other levels causing the samples to be unrepresentative (Everett, et al., 1976; Fenn, et al., 1977).

Total Well Construction Costs--

While general data have been presented for drilling and casing costs, additional costs such as for screens, gravel

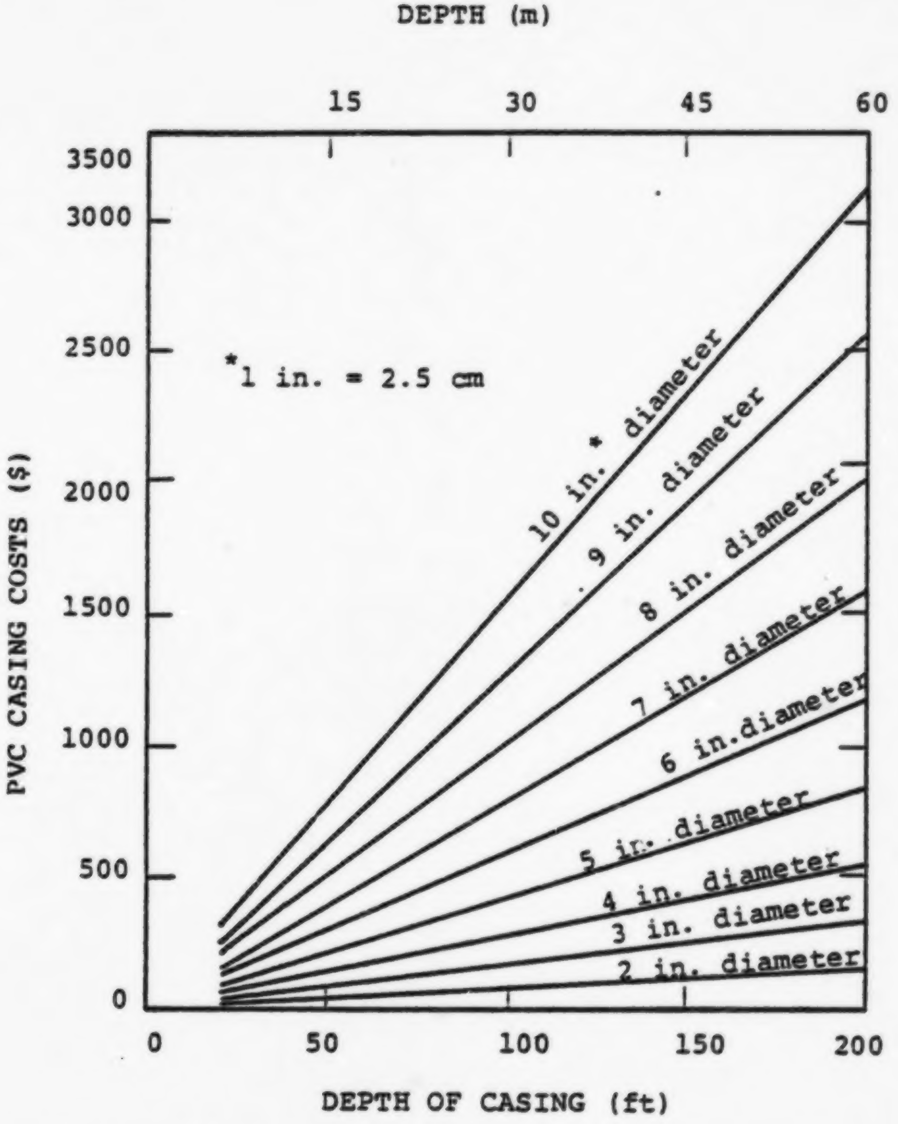


Figure 69. Costs of PVC casing, December, 1981, (after Everett, et al., 1976).

packing, grouting, well clean-up, etc. are highly dependent on local conditions. It is therefore, appropriate to provide separate cost data for completed wells in consolidated and unconsolidated formations. Total well construction costs (1981 dollars) based on 1974 costs (Everett, et al., 1976) for consolidated and unconsolidated formations are presented in Figures 70 and 71, respectively.

Monitoring Well Installations--

Sampling equipment is designed to meet specific monitoring requirements. The applicability and costs of single screened wells,, well clusters, and air-lift sampler wells are discussed in the following sub-sections. The advantages and disadvantages of each well system are summarized in Table 72.

Single screen wells--A PVC well slotted at a designated level within an aquifer constitutes the most common ground-water sampling approach (Figure 72). A serious limitation of this method; however, is its inability to provide a representative vertical distribution within the aquifer due to vertical mixing within the well casing. Point source sampling at discrete depths is a desirable characteristic which presently is not obtainable with a single screen well (Morrison and Brewer, 1981).

Costs of single screen wells for diameters ranging from 5 to 15 cm (2 to 6 in.) and installed depth of 3 to 61 m (10 to 200 ft), are presented in Figure 73. Costs include PVC casing, PVC slotted well monitoring screen, and bentonite seals. See Figures 67 and 68 for drilling costs.

Well clusters--A variety of devices have been proposed which are capable of sampling at predesignated levels within an aquifer. These approaches are inherently more useful than a single screened well, especially if only a single boring is required. One approach to the problem is the placement of several small-diameter wells within a single boring (Figure 74). Each cluster consists of a group of spaced small-diameter wells installed at different depths. These wells provide for water samples that are representative of the different levels within the aquifer. Judicious placement of well clusters at a landfill site and the surrounding area will allow delineation of both vertical and areal distribution of contaminants (Fenn, et al., 1977; Morrison and Brewer, 1981). Costs of cluster wells depend on bore-hole drilling costs, the number and diameter of the wells in each cluster, and the depth of each monitoring well. Drilling costs (Figures 67 and 68) and monitoring well costs (Figure 73) can be used to develop cluster well costs for site-specific situations.

Air-lift samplers--Air-lift samplers utilized for monitoring in the zone of saturation generally refer to any method in which

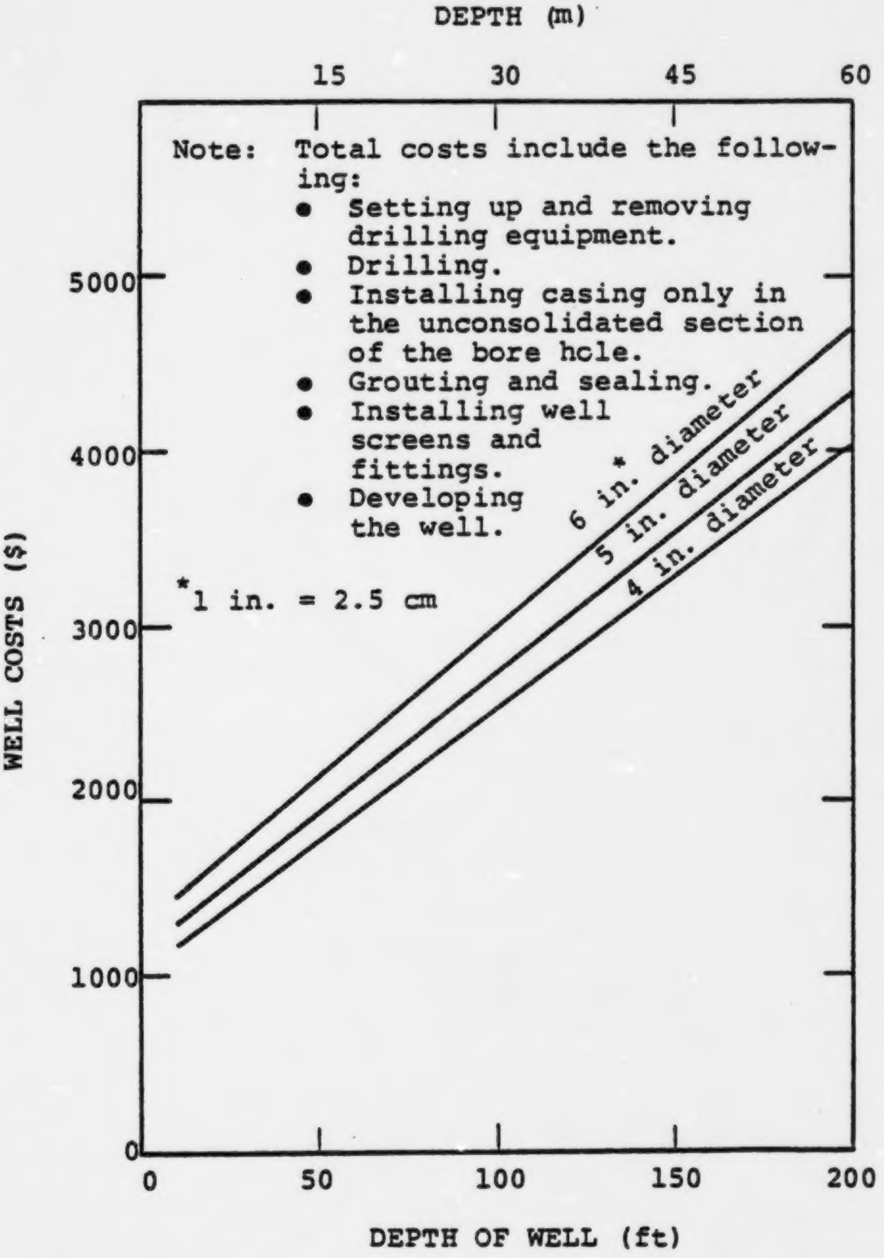


Figure 70. Total costs of 4-, 5-, and 6-inch diameter wells in consolidated formations, December, 1981 (after Everett, et al., 1976).

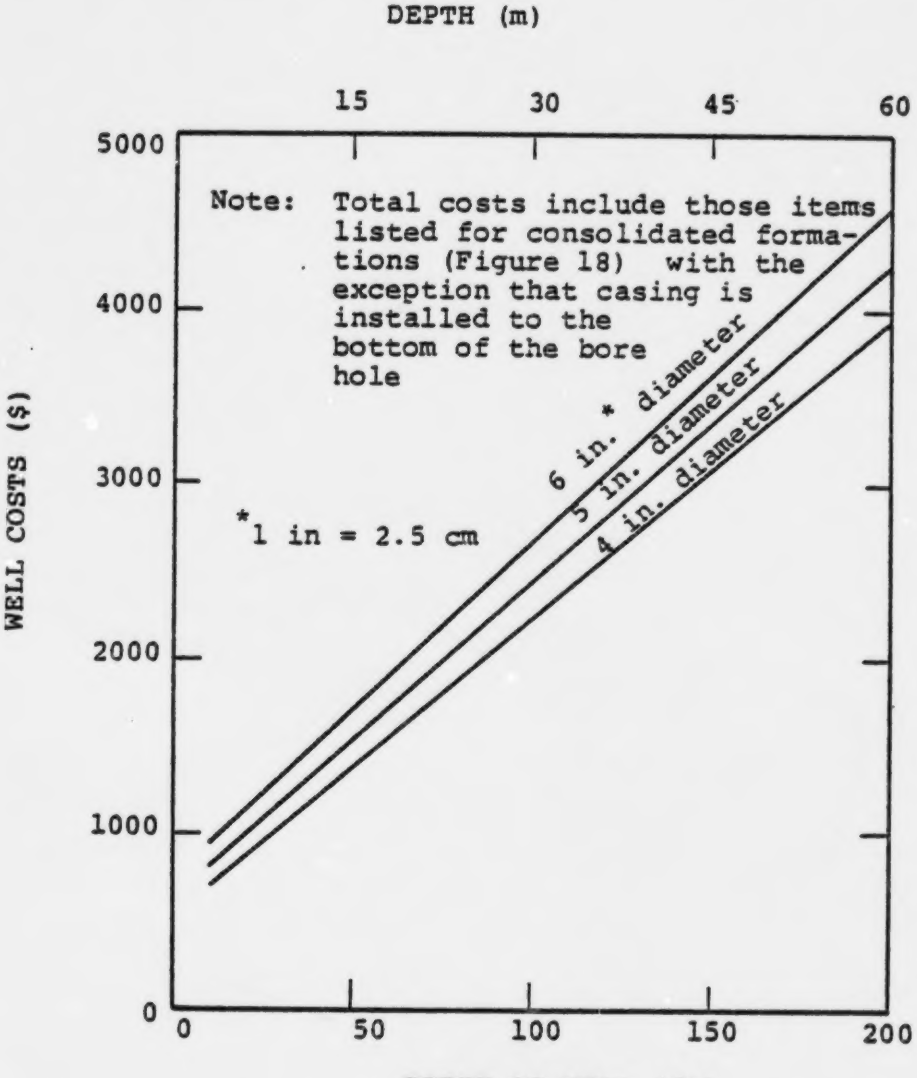


Figure 71. Total costs of 4-, 5-, and 6-inch diameter wells in unconsolidated formation, December, 1981 (after Gibb, 1971).

TABLE 72. ADVANTAGES AND DISADVANTAGES OF ZONE OF SATURATION MONITORING DEVICES (Everett,et al., 1976; Fenn,et al., 1977; Wilson, 1980; Morrison and Brewer, 1981)

Monitoring Device	Advantages	Disadvantages
Single Screened Wells	Shallow wells are cost effective and provide easy installation. Variety of drilling methods are available for installation. Easy to repair if damaged.	Provides samples from only one depth. Improper screening may provide erroneous results. Excessive screening may contribute to vertical leachate movement.
Multi-Probe Wells (Well Clusters)	Detailed vertical distribution of a contaminant may be obtained. Established methodology for vertical and areal distribution of contaminants. Easily installed.	Difficult to repair (e.g., single well, multi-part configuration). Sealing between screens may be difficult for multi-cluster wells. Proper well construction and sampling procedures are required for successful application.
Air-Lift Samplers	Sample collected from a specific section of the aquifer. Relatively inexpensive and easily installed if the casing diameter is small. Provides vertical sampling of the aquifer if properly engineered.	Difficult to repair if damaged. Improper placement of the device may cause problems in contaminant determination. Tubing susceptible to clogging or breakage.

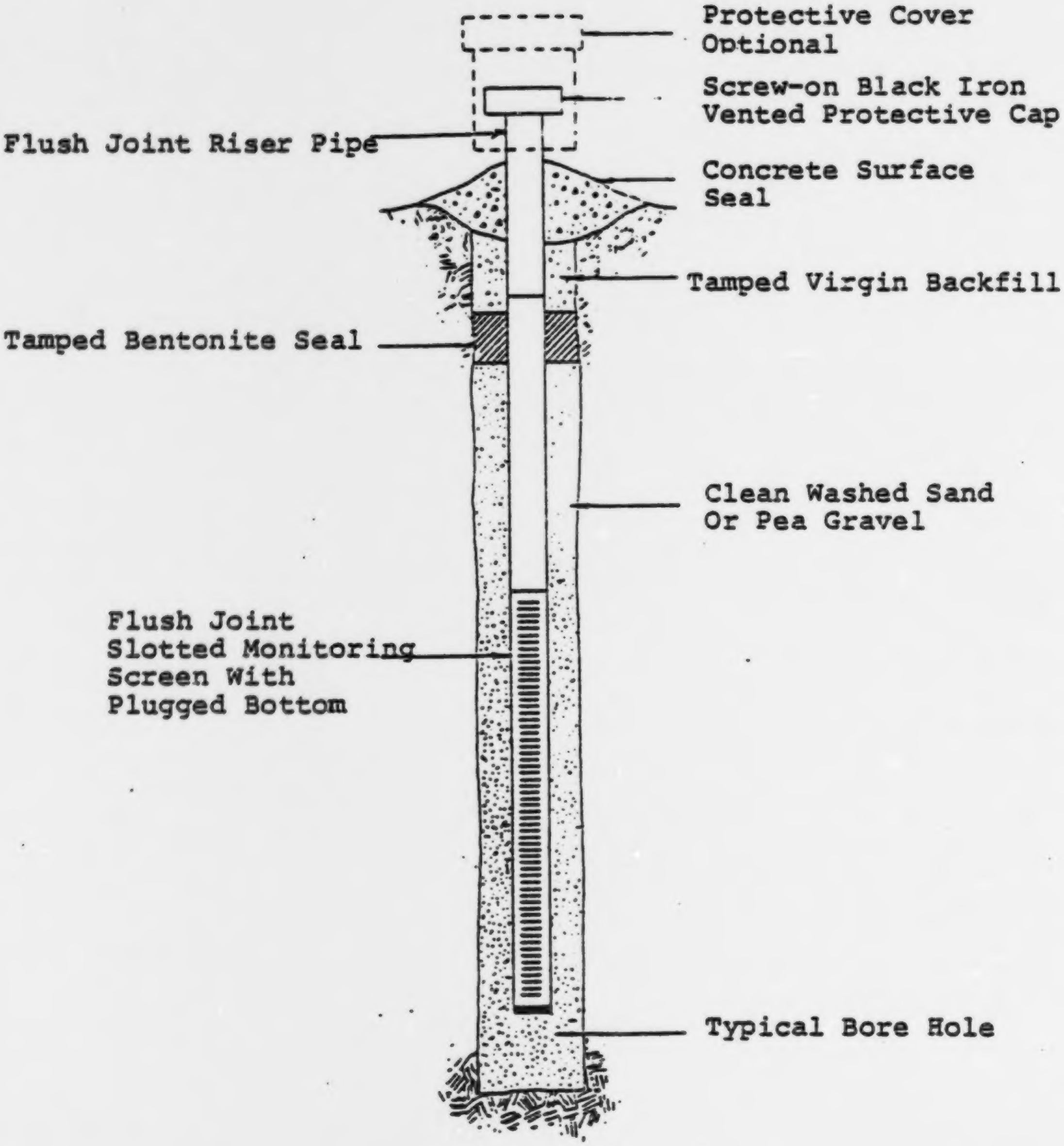

Protective Cover Optional
Screw-on Black Iron Vented Protective Cap
Flush Joint Riser Pipe
Concrete Surface Seal
Tamped Virgin Backfill
Tamped Bentonite Seal
Clean Washed Sand Or Pea Gravel
Flush Joint Slotted Monitoring Screen With Plugged Bottom
Typical Bore Hole

Figure 72. Single screened groundwater monitoring well (Timco, 1980).

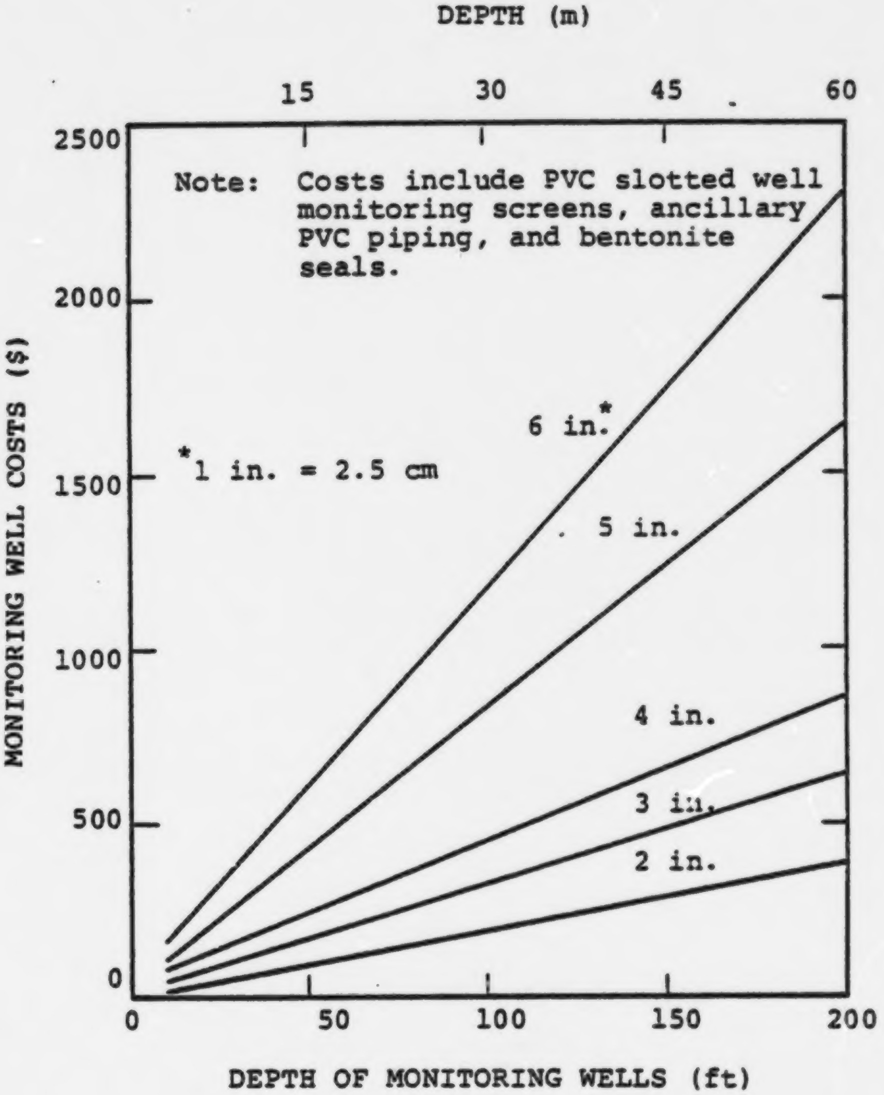


Figure 73. Costs of 2-, 3-, 4-, 5-, and 6-inch diameter monitoring wells, December, 1981 (after Timco Mfg. Co., Inc., 1980).

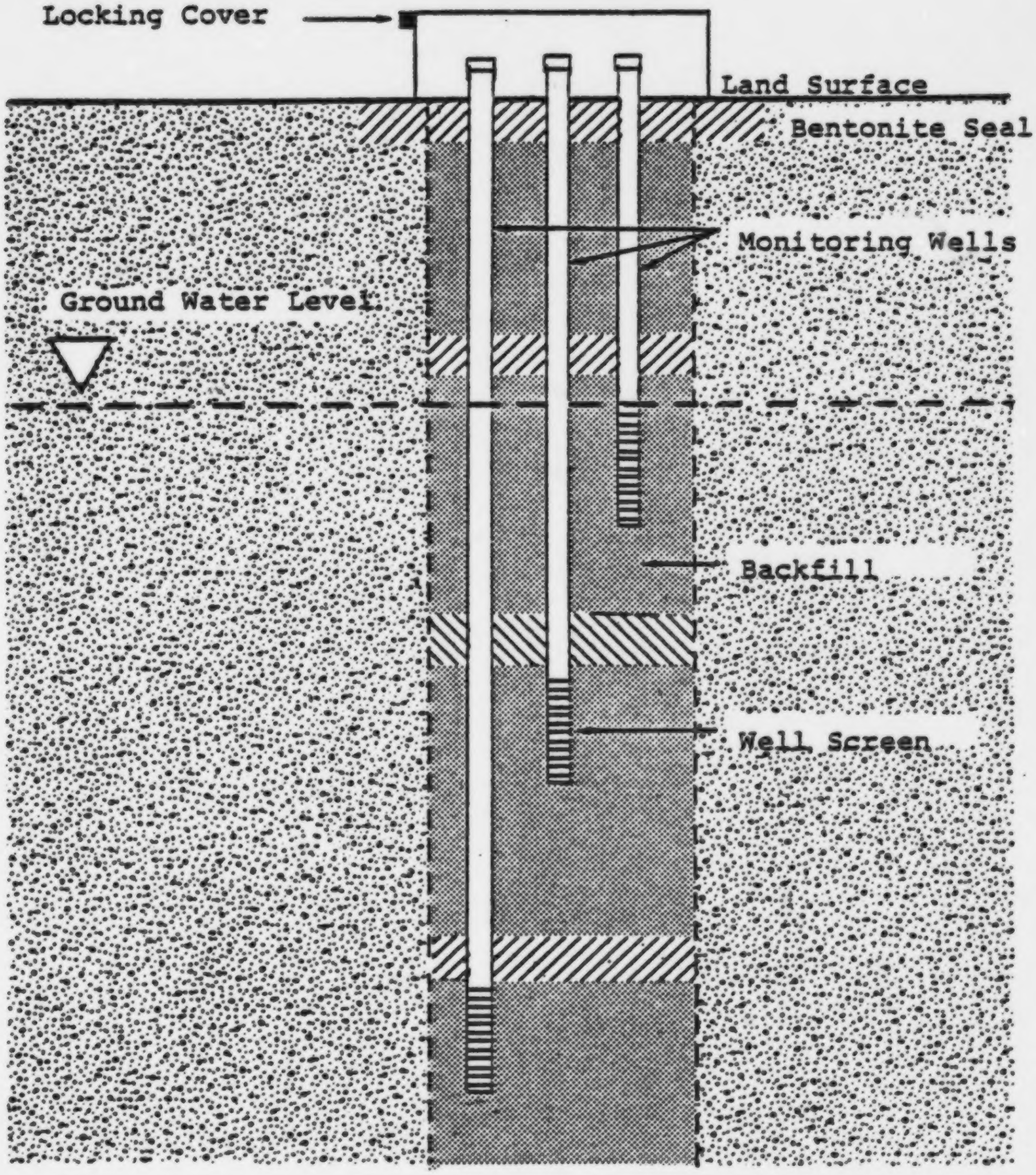

Locking Cover
Land Surface
Bentonite Seal
Monitoring Wells
Ground Water Level
Backfill
Well Screen

Figure 74. Well clusters (Morrison and Brewer, 1981).

a pressurized carrier gas is used to transport a sample to the surface. While a variety of designs exist, a check-valve arrangement, which allows water entry and closes when a carrier gas is injected into the sampler, is a common feature of these devices. The ability to collect samples at various depths within a single boring is a valuable characteristic of an air-lift sampler; many investigators have employed air-lift samplers exclusively for this purpose (Morrison and Brewer, 1981).

The installation of air-lift samplers consists of two approaches. The selection depends upon whether one sampler or a series of samplers is placed within the boring. In the case where one sampler is used (Figure 75), the sampler can be threaded onto a well casing and placed at the desired depth. The input/output tubes are situated within the well casing and lead to the surface. Costs of a single air-lift sampler for 3.2 cm (1¼ in.), 3.8 cm (1.5 in.) and 5.1 cm (1¼ in.) diameters and installed depths of 3 to 61 m (10 to 200 ft), are presented in Figure 76. Costs include slotted PVC air-lift sampler, PVC casing and collection tubing, and bentonite seals. See Figures 67 and 68 for drilling costs.

Where several air-lift samplers are required in a vertical section, the casing is omitted and the tubing collected at the surface as shown in Figure 75. This array provides greater flexibility, especially in three dimensional, volume monitoring. Costs of multiple air-lift samplers depend on the diameter and number of installed units, and the depth of the single boring. Drilling costs (Figures 67 and 68) and air-lift sampler costs (Figure 76) can be used to develop multiple air-lift sampler costs for site-specific situations.

MONITORING APPROACHES AND CONSIDERATIONS

Selection of Sampling Locations

Vadose Zone--

Monitoring within the vadose zone underlying a landfill site affords an early warning of potential ground-water pollution. If remedial measures are implemented prior to the onset of pollution, the associated renovation costs will be eliminated or reduced. Concomitantly, vadose zone monitoring may reduce the need for extensive ground-water monitoring, i.e., if a vadose zone monitoring program fails to detect the movement of contaminants, the requirements for ground-water monitoring may be reduced or largely precluded (Wilson, 1981). The savings in costs for constructing ground-water wells could be significant, particularly in western regions where water tables are often hundreds of feet deep. Because of the limited use of vadose samplers; however, little literature concerning their placement is available. For this reason, general guidelines are presented for the vadose zone.

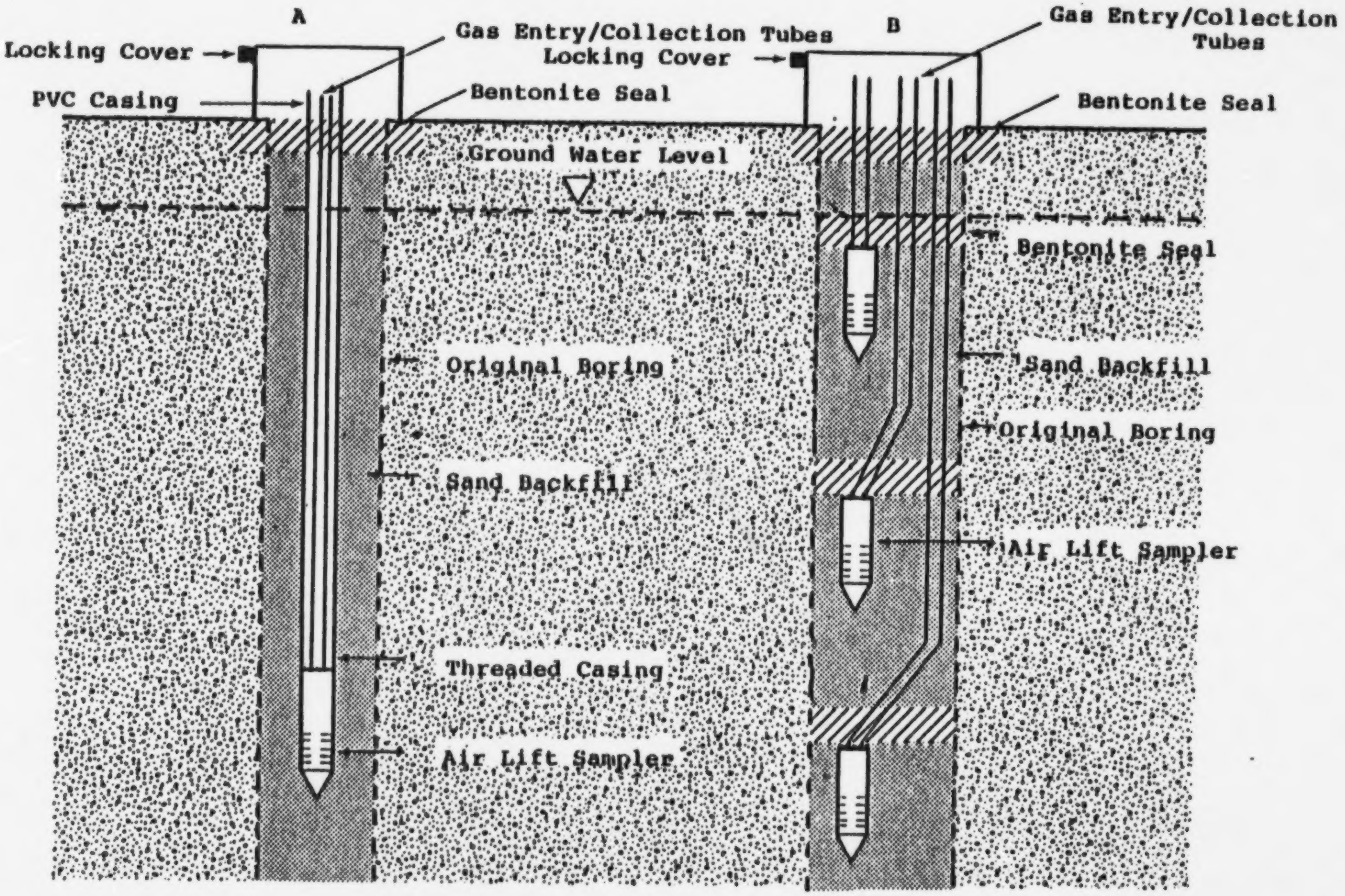

A
Locking Cover
PVC Casing
Gas Entry/Collection Tubes
Bentonite Seal
Ground Water Level
Original Boring
Sand Backfill
Threaded Casing
Air Lift Sampler
B
Locking Cover
Gas Entry/Collection Tubes
Bentonite Seal
Bentonite Seal
Sand Backfill
Original Boring
Air Lift Sampler

Figure 75. Single (A) and multiple (B) installation configurations for air-lift samplers. (Morrison and Brewer, 1981).

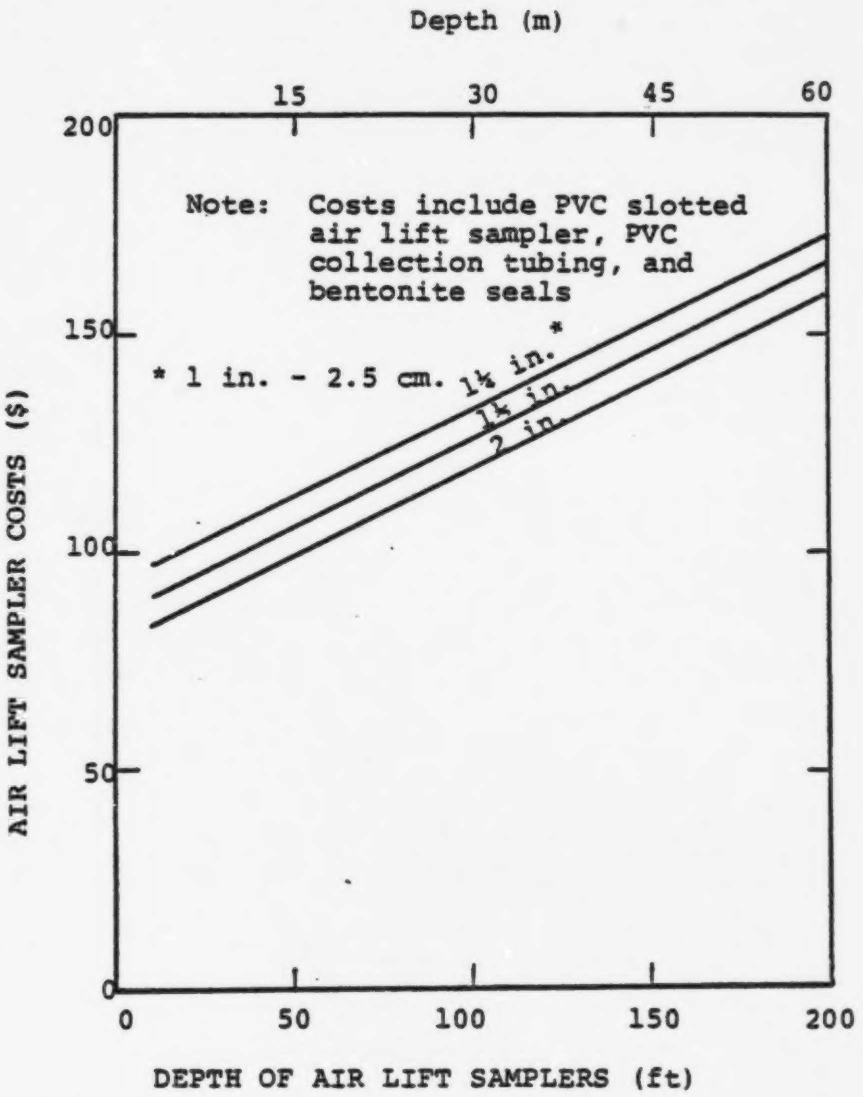


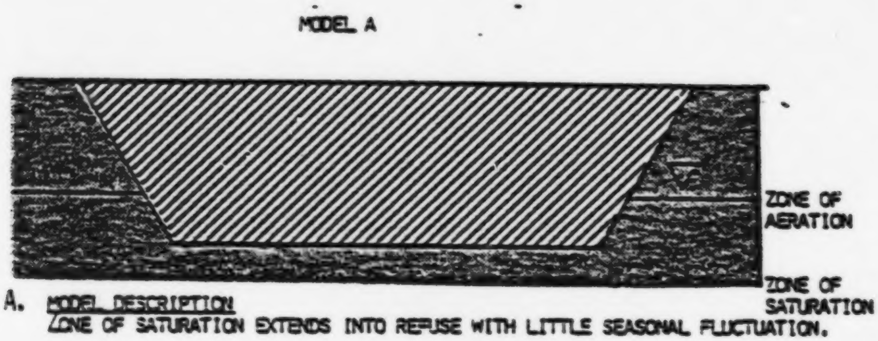
Figure 76. Costs of 1¼-, 1½-, and 2-inch diameter air lift samplers, December, 1981 (after Timco Mfg. Co., Inc. 1980).

Selection of an optimum location for sampling and non-sampling devices within the zone of aeration is largely site-specific. Figure 77 was developed for several hypothetical ground-water situations which may be encountered at disposal facilities and the corresponding placement of vadose monitoring devices within each system. In all cases, the water collection devices are situated below the landfill rather than within the refuse because of the need for developing and maintaining contact between the interstitial waters and the porous ceramic cup. In models D and E, non-sampling devices are also included in the array. In model D, these devices may be used to identify optimum periods for water collection. In model E, the non-sampling devices and any ground-water wells determine the exact depth to the zone of saturation. The vadose zone sampling devices may actually be employed to collect samples which are within the zone of saturation. Figure 78 illustrates an actual monitoring program as described by Johnson and Cartwright (1978) at the Genesco landfill in Illinois, in which vadose samplers and ground-water wells were successfully used in an integrated system.

Zone of Saturation--

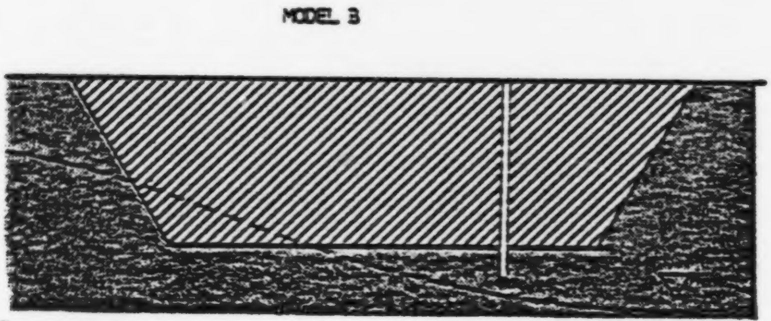
A monitoring program is established to give a prompt indication of ground-water contamination. The size of the landfill, hydrogeologic environment, and budgetary constraints are factors which will dictate the actual number of installed wells. The spacing and depths of monitoring wells depends on the particular pattern of ground-water flow, making it extremely difficult to specify national minimums or maximums in this area. It is recommended that the following minimal monitoring well network be implemented in order to detect and evaluate potential ground-water contamination at a sanitary landfill (Fenn, *et al.*, 1977; EPA, 1980).

- One well upgradient from the landfill to characterize ground-water quality in the uppermost aquifer. The owner or operator must assure that the upgradient samples represent true background conditions and are not contaminated by the facility;
- One well placed adjacent to the downgradient waste boundary to give a prompt indication of ground-water contamination; and
- A minimum of three wells downgradient of the landfill to provide representative samples capable of detecting migration of contaminants from the facility. Once contamination is detected, additional lines of wells can be constructed downgradient to gauge dispersion and attenuation of the leachate, thereby providing the necessary information for predicting the ultimate



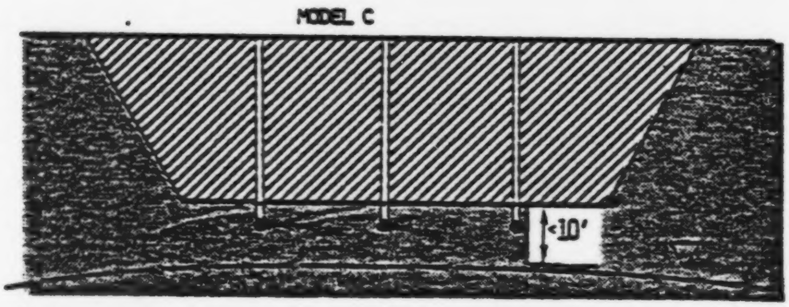
MODEL A

A. MODEL DESCRIPTION
ZONE OF SATURATION EXTENDS INTO REFUSE WITH LITTLE SEASONAL FLUCTUATION.



MODEL B

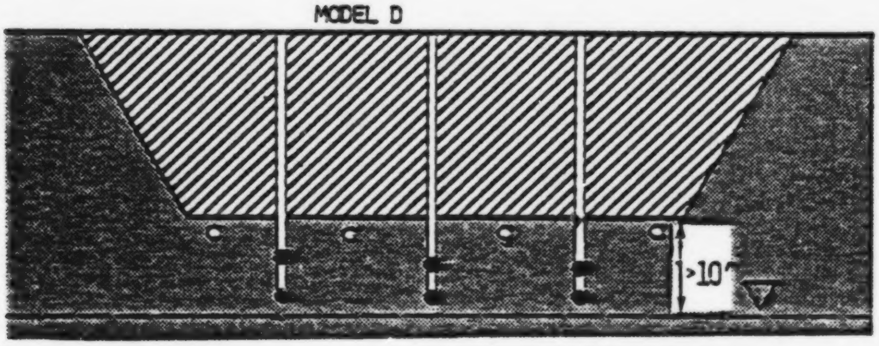
B. MODEL DESCRIPTION
ZONE OF SATURATION EXTENDS PARTIALLY INTO REFUSE WITH LITTLE SEASONAL FLUCTUATION.



MODEL C

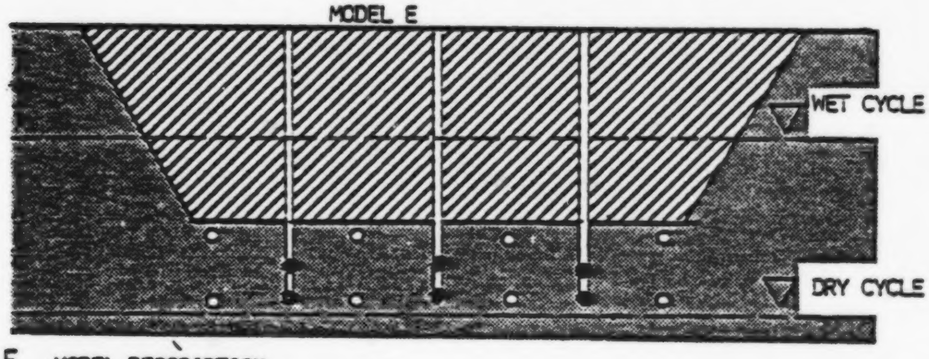
C. MODEL DESCRIPTION
ZONE OF SATURATION BELOW REFUSE/SOIL INTERFACE: ZONE OF AERATION EXTENDS <10' BELOW REFUSE/SOIL INTERFACE

Figure 77. Hypothetical hydrologic conditions and possible location of vadose monitoring.



D. MODEL DESCRIPTION

IDENTICAL TO MODEL C EXCEPT FOR DEPTH OF ZONE OF AERATION BELOW THE REFUSE/SOIL INTERFACE ASSUMED >10'



E. MODEL DESCRIPTION

DEPTH TO ZONE OF SATURATION FLUCTUATES SIGNIFICANTLY

LEGEND:

SANITARY LANDFILL	▨
GROUNDWATER LEVEL	▽
SAMPLING DEVICE (LYSIMETER)	●
NON-SAMPLING DEVICE (MOISTURE BLOCK, ETC.)	○

Figure 77. (Continued).

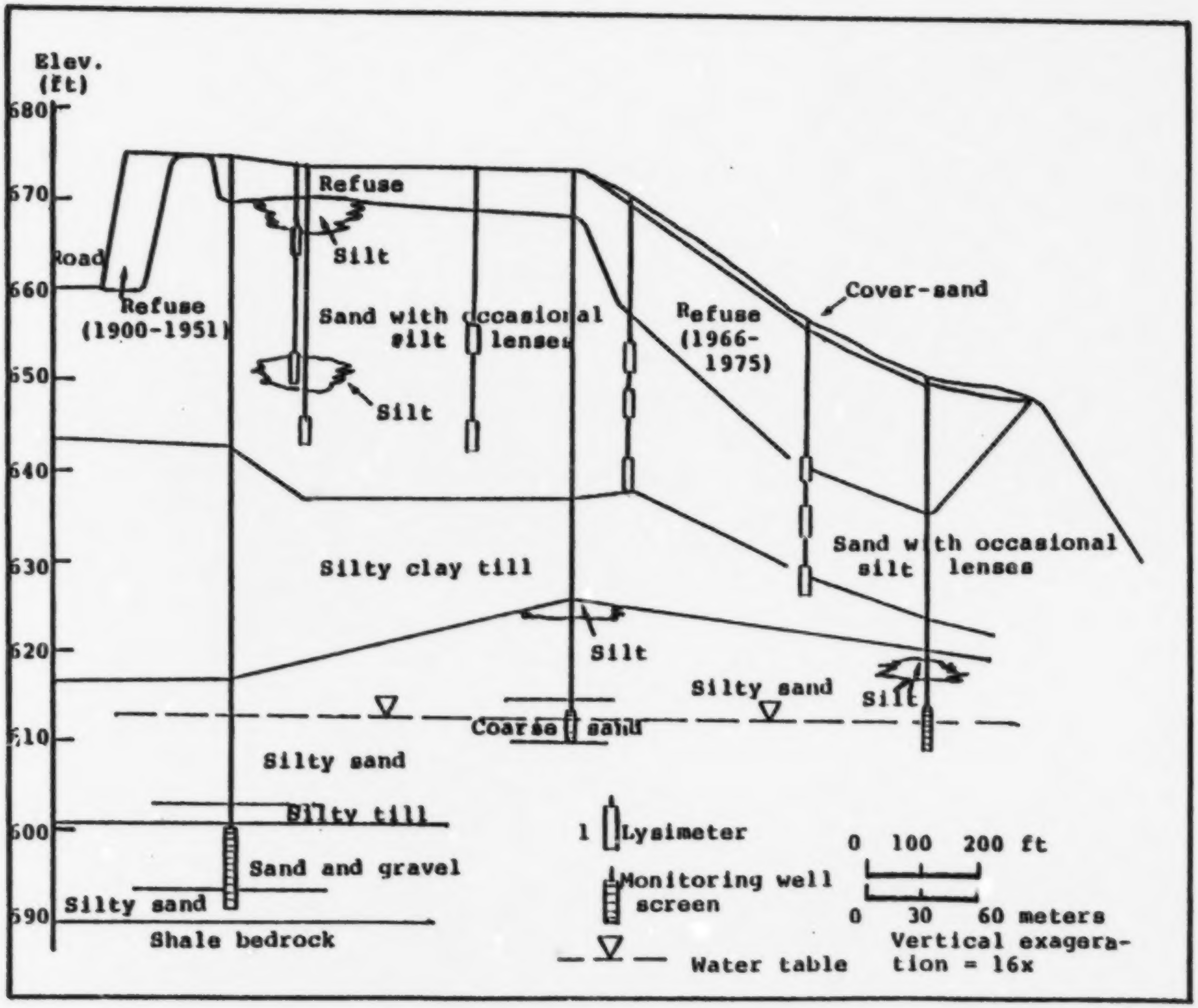

Elev. (ft)
680
670
660
650
640
630
620
610
600
590
Road
Refuse (1900-1951)
Refuse
Silt
Sand with occasional silt lenses
Silt
Refuse (1966-1975)
Cover-sand
Sand with occasional silt lenses
Silty clay till
Silt
Silty sand
Silt
Coarse sand
Silty sand
Silty till
Sand and gravel
Silty sand
Shale bedrock
Lysimeter
Monitoring well screen
Water table
0 100 200 ft
0 30 60 meters
Vertical exageration = 16x

Figure 78. Cross section of Geneseo Landfill and underlying materials showing a portion of the monitoring well-lysimeter network (Johnston and Cartwright, 1978).

fate of the plume. This approach will be an expensive and time-consuming process, the necessity for which will depend on regulatory requirements.

Monitoring Frequency

Monitoring frequency is greatly influenced by many factors, including (Fenn, et al., 1977):

- Purpose for monitoring (e.g., compliance versus investigatory; routine versus synoptic);
- Background quality of ground water at the site;
- Characteristics of ground-water flow;
- Monitoring network utilized;
- Location and purpose of the particular monitoring well;
- Other sources of ground-water contamination;
- Trends in the monitoring data;
- Legal and institutional needs;
- Climatological characteristics; and
- Site history.

The monitoring frequency should be flexible, allowing for modification to meet the individual needs of each site. One useful monitoring approach combines synoptic and routine sampling. The synoptic sampling provides data during events representing the greatest pollution potential (e.g., heavy precipitation, snow melt, flooding). The scheduled sampling measures seasonal and other long term phenomena. Because of site-specific requirements, it is not possible to delineate national maximums or minimums in this area.

Monitoring Parameters

The parameters selected for study will be determined largely by the purpose of the monitoring program. Many researchers have developed lists of key indicator parameters (Goerlitz and Brown, 1972; EPA, 1973; EPA 1975; Fenn, et al., 1977; EPA, 1979, EPA, 1980). Such lists can be referred to for the purpose of tailoring site-specific monitoring programs. EPA (1980) established guidelines for ground-water monitoring of three sets of parameters, each of which serve a different purpose. These key parameters may be modified, as required, to meet site-specific situations. The three sets of parameters are summarized as follows:

1. The first set reflects the aquifer's suitability as a drinking water supply. These parameters, are those specified in the Interim Primary Drinking Water Regulations established under the Safe Drinking Water Act. The purpose of the initial sampling for drinking water parameters is to identify facilities that may be severely degrading present and future drinking water supplies. It should be noted that the specified parameters do not comprise a complete list of contaminants that define an aquifer's potential as an acceptable drinking water supply; however, these constituents will be useful in establishing monitoring priorities;

2. The second set of parameters includes chloride, iron, manganese, phenols, sodium, and sulfate. These parameters are generally recognized as useful for characterizing ground-water quality. These contaminants are ubiquitous in the environment and are often used to characterize a ground-water supply's suitability for a variety of uses. Information on these parameters will be useful in any assessment of ground-water contamination that follows the determination that a facility is leaking. Such information will, for example, assist in determining the extent to which contamination of the aquifer may be coming from sources other than the disposal facility; and

3. The third set of parameters consists of four indicators that will be used to determine whether a facility is leaking. It is important for a facility to answer the threshold question of whether hazardous waste constituents are entering the aquifer underlying the facility. The four indicators--specific conductance, pH, total organic carbon, and total organic halogen--reflect changes in the organic and inorganic makeup of the ground water. A statistically significant change (increase or decrease for pH, increase only for the others) in these indicators between the initial background concentration or value and those from downgradient wells suggests that organic or inorganic substances are being introduced into the aquifer by the facility. Increases in specific conductance indicate the presence of inorganic substances in

the ground water. Likewise, increases or decreases in pH suggest the presence of inorganic contamination. Total organic carbon (TOC) and total organic halogen (TOX) concentrations in ground water tend to increase as a result of organic contributions from a solid waste facility. The methodology to sample and analyze for these indicators is presently available. Monitoring these indicators will be sufficient to make the threshold assessment of whether a facility is leaking.

Sample Collection Methods; Containers and Sample Preservation, and Analytical Procedures.

Sample Collection--

Either a vacuum or pressure device may be used to withdraw samples from lysimeters (Wood, 1973). For lysimeters placed deeper than 20 ft, pressure devices utilizing nitrogen as a carrier gas have been found effective (Morrison and Brewer, 1981). For non-sampling devices, differences in the moisture content may be recorded continously or at selected times.

The method chosen for sampling within the zone of saturation depends on the depth of the water table. A suction-lift pump is installed above a well. The maximum suction lift is limited by atmospheric pressure, head losses due to friction, and the required inlet head of the pump. At sea level, the best designed pumps usually achieve a suction lift of 7.6 m (25 ft), while the suction lift of an average pump varies from 4.6 to 5.5 m (15 to 18 ft) (Everett, et al., 1976). Where the depth to the water table is greater than the practical suction limit, peristalic, centrifugal,or submersible pumps may be employed.

Submersible pumps require a minimum casing diameter of 10 cm (4 in.). A submersible pump is feasible where a number of monitoring wells are located in a small area and only periodic pumping is required (Everett, et al., 1976). Submersible pumps are considered the best, but most expensive, method of collecting water samples (Fenn, et al.,1977). McMillian and Keeley (1968) reported the use of a truck portable submersible pump capable of withdrawing water samples at depths up to 91 m (300 ft) inside a casing as small as 11.4 cm (4½ in.). With this equipment, cross-contamination between wells from the sampling equipment will be virtually eliminated by the flushing action during pre-pumping.

Water samples may also be taken by bailing. Where pumping equipment is not available, or where a pump cannot be installed because the diameter of the well is too small, a container such as a weighted bottle or a short section of pipe capped at the bottom can be lowered into the well to collect a sample. This sample will give an indication of the chemical quality of the water in the aquifer, but should not be used for bacteriological or detailed chemical analysis because of the likelihood of contamination. This method is not recommended except where other methods of sampling are unavailable (Everett, et al., 1976).

Air lift sampling can be useful in monitoring wells that need to be pumped only at periodic intervals. The advantage of this sampler is that the apparatus can be designed for permanent installation in the monitoring well. The air-lift sampler eliminates the possibility of cross-contamination between wells as can occur with portable pumping and sampling devices. This method is also of value in obtaining bacteriological samples where external sources of contamination must be avoided (Fenn, et al., 1977). Johnson Division UOP (1971) presents data on the relation between total lift, submergence of air lines, and water discharged per cubic foot of air delivered by an air compressor.

Where it is desired to collect water from a specific depth within a well, a special cylinder known as a "thief sampler," can be lowered by cable into the well and closed at a predetermined depth. Most types consist of a cylinder that can be less than 5 cm (2 in.) in diameter. A bar with a plunger or cork at each end runs through the cylinder. When the sampler is lowered below the water table to the desired depth, the two ends are closed. The most common type of "thief sampler" is spring loaded. With this type, a messenger is lowered along the cable to release the two end cups, which in turn trap the water sample. Many "thief samplers" are custom made for a specific use (Everett, et al., 1976). Additional detailed descriptions of the operation and application of different types of ground-water samplers are available in the literature (Cherry, 1965; Trescott and Pinder, 1970; Sommerfeldt and Campbell, 1975).

Containers and Sample Preservation--

EPA (1979) and APHA (1981) provide a detailed presentation of appropriate containers for various chemical species and sample preservation techniques. Methods of preservation are relatively limited and are intended to:

(1) Retard biological action;

(2) Retard hydrolysis of chemical compounds and complexes; and

(3) Reduce volatility of constituents.

Preservation methods are generally limited to pH control, chemical addition, refrigeration, and freezing. Recommended containers, preservatives, and holding times for various samples are given in Table 73 (EPA, 1979).

TABLE 73. RECOMMENDATIONS FOR SAMPLING AND PRESERVATION OF SAMPLES (EPA, 1979).

Measurement	Sample Volume Required (ml)	Container	Preservative	Holding Time
Acidity	100	P[*], G[†]	Cool, 4°C	24 Hrs.
Alkalinity	100	P, G	Cool, 4°C	24 Hrs.
Arsenic	100	P, G	HNO_3 to pH<2	6 Mos.
BOD	1000	P, G	Cool, 4°C	6 Hrs.
Bromide	100	P, G	Cool, 4°C	24 Hrs.
COD	50	P, G	H_2SO_4 to pH<2	7 Days
Chloride	50	P, G	None Req.	7 Days
Color	50	P, G	Cool, 4°C	24 Hrs.
Cyanides	500	P, G	Cool, 4°C NaOH to pH 12	24 Hrs.
Dissolved Oxygen				
Probe	300	G only	Det. on site	No Holding
Winkler	300	G only	Fix on site	4-8 Hrs.
Fluoride	300	P, G	Cool, 4°C	7 Days
Hardness	100	P, G	Cool, 4°C HNO_3 to pH<2	7 Days
Iodide	100	P, G	Cool, 4°C	24 Hrs.
MBAS	250	P, G	Cool, 4°C	24 Hrs.
Metals				
Dissolved	200	P, G	Filter on site HNO_3 to pH<2	6 Mos.
Suspended			Filter on site	6 Mos.
Total	100		HNO_3 to pH<2	6 Mos.
Mercury				
Dissolved	100	P, G	Filter HNO_3 to pH<2	38 Days (Glass) 13 Days (Hard Plastic)

(continued)

TABLE 73 (continued)

Measurement	Sample Volume Required (ml)	Container	Preservative	Holding Time
Total	100	P, G	HNO_3 to pH<2	38 Days (Glass) 13 Days (Hard Plastic)
Nitrogen				
Ammonia	400	P, G	Cool, 4°C H_2SO_4 to pH<2	24 Hrs
Kjeldahl, total	500	P, G	Cool, 4°C H_2SO_4 to pH<2	7 Days
Nitrate	100	P, G	Cool, 4°C H_2SO_4 to pH<2	24 Hrs.
Nitrite	50	P, G	Cool, 4°C	24 Hrs.
NTA	50	P, G	Cool, 4°C	24 Hrs.
Oil & Grease	1000	G only	Cool, 4°C H_2SO_4 or HCl to pH<2	24 Hrs.
Organic Carbon	25	P, G	Cool, 4°C H_2SO_4 to pH<2	24 Hrs.
pH	25	P, G	Cool, 4°C Det. on site	6 Hrs.
Phenolics	500	G only	Cool, 4°C H_3PO_4 to pH<4 1.0 g $CuSO_4$/l	24 Hrs.
Phosphorus				
Ortho-phosphate, Dissolved	50	P, G	Filter on site Cool, 4°C	24 Hrs.
Hydrolyzable	50	P, G	Cool, 4°C H_2SO_4 to pH<2	24 Hrs.
Total	50	P, G	Cool, 4°C	7 Days
Total, Dissolved	50	P, G	Filter on site Cool, 4°C	24 Hrs.
Residue				
Filterable	100	P, G	Cool, 4°C	7 Days

(continued)

TABLE 73. (concluded)

Measurement	Sample Volume Required (ml)	Container	Preservative	Holding Time
Non-Filterable	100	P, G	Cool, 4°C	7 Days
Total	100	P, G	Cool, 4°C	7 Days
Volatile	100	P, G	Cool, 4°C	7 Days
Settleable Matter	1000	P, G	None Req.	24 Hrs.
Selenium	50	P, G	HNO_3 to pH<2	6 Mos.
Silica	50	P only	Cool, 4°C	7 Days
Specific Conductance	100	P, G	Cool, 4°C	24 Hrs.
Sulfate	50	P, G	Cool, 4°C	7 Days
Sulfide	500	P, G	2 ml zinc acetate	24 Hrs.
Sulfite	50	P, G	Det. on site	No Holding
Temperature	1000	P, G	Det. on site	No Holding
Threshold Odor	200	G only	Cool, 4°C	24 Hrs.
Turbidity	100	P, G	Cool,4°C	7 Days

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[*] P = plastic (polyethylene or equivalent)

[†] G = glass

Analytical Procedures and Costs--

EPA (1979) and APHA (1981) describe detailed analytical procedures which, in general, are quite comparable. The methods are applicable to both water and wastewaters, and both fresh and saline water samples. Tests procedures are given for the measurement of physical, inorganic, and selected organic constituents and parameters. Instrumental methods have been selected in preference to manual procedures because of the improved speed, accuracy, and precision. Precision and accuracy statements are provided where such data are available.

Analytical costs vary considerably in the U.S. The required sensitivity, degreee of accuracy, number of samples, time constraints, and laboratory location are major considerations. Water quality analyses costs (1981 dollars) for a commercial laboratory are given in Table 74 (West Coast Technical Services, Inc., 1981).

SUMMARY

Monitoring equipment for the vadose zone may be classified as sampling or non-sampling. Sampling approaches include pore-water extraction from soil cores and deployment of pressure/vacuum lysimeters to obtain in situ water samples for chemical analyses. Non-sampling equipment includes tensiometers, psychrometers, electrical resistance blocks, and neutron moisture logging. Non-sampling methods provide for the determination of water content and water movement in the vadose zone.

A zone of saturation monitoring program is established to give a prompt indication of ground-water contamination. The size of the landfill, hydrogeologic environment, and budgetary constraints are factors which will dictate the actual number of wells. The spacing and depths of monitoring wells depends on the particular pattern of ground-water flow. Monitoring within the zone of aeration has traditionally been accomplished with single screened wells. More recently developed installations such as the cluster well and the air-lift sampler well allows for multiple sampling points throughout the aquifer. Multi-sampling wells provide for the vertical distribution of contaminants in the aquifer. This information is of great value for the determination of leachate migration and changes in ground-water quality.

Monitoring frequency should be flexible, allowing for modification to meet the individual needs of each site. One useful monitoring approach combines synoptic and routine sampling. Monitoring parameters selected for study will be determined largely by the purpose of the monitoring program. Many researchers have developed lists of key indicator parameters. The selection of key monitoring parameters should be modified, as required, to meet site-specific situations.

TABLE 74. COSTS OF WATER QUALITY ANALYSES, DECEMBER, 1981.
(West Coast Technical Service, Inc., 1981)

	Price per sample		
Priority Pollutants	Wastewater		Sludge
Organic	$825		$1080
Metals (Be, Cd, Cr, Cu, Pb, Ni, Ag, Th, Zn, Sb, As, Se, Hg)	430		500
	Samples submitted at one time (price per sample)		
	1-3	4-7	8+
Pesticides and Herbicides by GC & LC			
Organochlorine	$120	$100	$ 75
Triazine	120	100	75
Organophosphorous	130	110	85
Fluralins	120	100	75
Carbamate and Urea	180	145	130
Polychlorinated Biphenyls (PCB's)			
In waste water	$150	$120	$100
In transformer oil	120	100	85
Haloethers	$120	$100	$ 75
Chlorinated Hydrocarbons	170	145	120
Phthalates	145	120	95
Polynuclear Aromatics (PNA's) by GCMS	220	195	185
Phenols by GCMS	200	180	170
Benzidines	240	210	180
Nitroaromatics and Isophorone	170	145	120
Nitrosamines	170	145	120
GCMS Confirmation when possible	180	180	180

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APPENDIX A. THORNTHWAITE TABLES

TABLE A-1. MONTHLY VALUES OF i CORRESPONDING TO MONTHLY MEAN TEMPERATURES (°C) *

T°C	.0	.1	.2	.3	.4	.5	.6	.7	.8	.9
0			.01	.01	.02	.03	.04	.05	.06	.07
1	.09	.10	.12	.13	.15	.16	.18	.20	.21	.23
2	.25	.27	.29	.31	.33	.35	.37	.39	.42	.44
3	.46	.48	.51	.53	.56	.58	.61	.63	.66	.69
4	.71	.74	.77	.80	.82	.85	.88	.91	.94	.97
5	1.00	1.03	1.06	1.09	1.12	1.16	1.19	1.22	1.25	1.29
6	1.32	1.35	1.39	1.42	1.45	1.49	1.52	1.56	1.59	1.63
7	1.66	1.70	1.74	1.77	1.81	1.85	1.89	1.92	1.96	2.00
8	2.04	2.08	2.12	2.15	2.19	2.23	2.27	2.31	2.35	2.39
9	2.44	2.48	2.52	2.56	2.60	2.64	2.69	2.73	2.77	2.81
10	2.86	2.90	2.94	2.99	3.03	3.08	3.12	3.16	3.21	3.25
11	3.30	3.34	3.39	3.44	3.48	3.53	3.58	3.62	3.67	3.72
12	3.76	3.81	3.86	3.91	3.96	4.00	4.05	4.10	4.15	4.20
13	4.25	4.30	4.35	4.40	4.45	4.50	4.55	4.60	4.65	4.70
14	4.75	4.81	4.86	4.91	4.96	5.01	5.07	5.12	5.17	5.22
15	5.28	5.33	5.38	5.44	5.49	5.55	5.60	5.65	5.71	5.76
16	5.82	5.87	5.93	5.98	6.04	6.10	6.15	6.21	6.26	6.32
17	6.38	6.44	6.49	6.55	6.61	6.66	6.72	6.78	6.84	6.90
18	6.95	7.01	7.07	7.13	7.19	7.25	7.31	7.37	7.43	7.49
19	7.55	7.61	7.67	7.73	7.79	7.85	7.91	7.97	8.03	8.10
20	8.16	8.22	8.28	8.34	8.41	8.47	8.53	8.59	8.66	8.72
21	8.78	8.85	8.91	8.97	9.04	9.10	9.17	9.23	9.29	9.36
22	9.42	9.49	9.55	9.62	9.68	9.75	9.82	9.88	9.95	10.01
23	10.08	10.15	10.21	10.28	10.35	10.41	10.48	10.55	10.62	10.68
24	10.75	10.82	10.89	10.95	11.02	11.09	11.16	11.23	11.30	11.37
25	11.44	11.50	11.57	11.64	11.71	11.78	11.85	11.92	11.99	12.06
26	12.13	12.21	12.28	12.35	12.42	12.49	12.56	12.63	12.70	12.78
27	12.85	12.92	12.99	13.07	13.14	13.21	13.28	13.36	13.43	13.50
28	13.58	13.65	13.72	13.80	13.87	13.94	14.02	14.09	14.17	14.24
29	14.32	14.39	14.47	14.54	14.62	14.69	14.77	14.84	14.92	14.99
30	15.07	15.15	15.22	15.30	15.38	15.45	15.53	15.61	15.68	15.76
31	15.84	15.92	15.99	16.07	16.15	16.23	16.30	16.38	16.46	16.54
32	16.62	16.70	16.78	16.85	16.93	17.01	17.09	17.17	17.25	17.33
33	17.41	17.49	17.57	17.65	17.73	17.81	17.89	17.97	18.05	18.13
34	18.22	18.30	18.38	18.46	18.54	18.62	18.70	18.79	18.87	18.95
35	19.03	19.11	19.20	19.29	19.36	19.45	19.53	19.61	19.69	19.78
36	19.86	19.95	20.03	20.11	20.20	20.28	20.36	20.45	20.53	20.62
37	20.70	20.79	20.87	20.96	21.04	21.13	21.21	21.30	21.38	21.47
38	21.56	21.64	21.73	21.81	21.90	21.99	22.07	22.16	22.25	22.33
39	22.42	22.51	22.59	22.68	22.77	22.86	22.95	23.03	23.12	23.21
40	23.30									

$$* \; i = \left(\frac{T}{5}\right)^{1.514}$$

TABLE A-2.

VALUES OF UNADJUSTED DAILY POTENTIAL EVAPOTRANSPIRATION (mm)
FOR DIFFERENT MEAN TEMPERATURES (°C) AND I VALUES

T°C \ I*	25.0	27.5	30.0	32.5	35.0	37.5	40.0	42.5	45.0	47.5	50.0	52.5
0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
.25	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
.50	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0
.75	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.0	0.0
1.00	0.2	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1
1.25	0.3	0.3	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1
1.50	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.1	0.1	0.1
1.75	0.4	0.4	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.1
2.00	0.4	0.4	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.2
2.25	0.5	0.5	0.4	0.3	0.3	0.3	0.3	0.3	0.3	0.2	0.2	0.2
2.50	0.5	0.5	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.3	0.3	0.2
2.75	0.6	0.6	0.5	0.4	0.4	0.4	0.4	0.4	0.3	0.3	0.3	0.3
3.00	0.6	0.6	0.5	0.5	0.5	0.4	0.4	0.4	0.4	0.3	0.3	0.3
3.25	0.7	0.7	0.6	0.5	0.5	0.5	0.4	0.4	0.4	0.4	0.4	0.3
3.50	0.7	0.7	0.6	0.6	0.5	0.5	0.5	0.5	0.4	0.4	0.4	0.4
3.75	0.8	0.7	0.6	0.6	0.5	0.5	0.5	0.5	0.5	0.4	0.4	0.4
4.00	0.8	0.8	0.7	0.6	0.6	0.6	0.5	0.5	0.5	0.5	0.4	0.4
4.25	0.9	0.8	0.7	0.7	0.6	0.6	0.6	0.6	0.5	0.5	0.5	0.4
4.50	0.9	0.9	0.8	0.7	0.7	0.6	0.6	0.6	0.6	0.5	0.5	0.5
4.75	1.0	0.9	0.8	0.8	0.7	0.7	0.6	0.6	0.6	0.5	0.5	0.5
5.00	1.0	1.0	0.9	0.8	0.8	0.7	0.7	0.7	0.6	0.6	0.5	0.5
5.25	1.1	1.0	0.9	0.9	0.8	0.8	0.7	0.7	0.7	0.6	0.6	0.6
5.50	1.1	1.1	1.0	0.9	0.9	0.8	0.8	0.7	0.7	0.6	0.6	0.6
5.75	1.2	1.1	1.0	1.0	0.9	0.9	0.8	0.8	0.7	0.7	0.7	0.6
6.00	1.2	1.2	1.1	1.0	0.9	0.9	0.9	0.8	0.8	0.7	0.7	0.7
6.25	1.2	1.2	1.1	1.0	1.0	0.9	0.9	0.8	0.8	0.7	0.7	0.7
6.50	1.3	1.2	1.1	1.1	1.0	1.0	0.9	0.9	0.8	0.8	0.8	0.7
6.75	1.3	1.3	1.2	1.1	1.0	1.0	1.0	0.9	0.9	0.8	0.8	0.8
7.00	1.4	1.3	1.2	1.1	1.1	1.0	1.0	1.0	0.9	0.9	0.8	0.8
7.25	1.4	1.4	1.3	1.2	1.1	1.1	1.0	1.0	1.0	0.9	0.9	0.9
7.50	1.5	1.4	1.3	1.2	1.2	1.1	1.1	1.0	1.0	0.9	0.9	0.9
7.75	1.5	1.4	1.4	1.3	1.2	1.2	1.1	1.1	1.0	1.0	1.0	0.9
8.00	1.6	1.5	1.4	1.3	1.3	1.2	1.2	1.1	1.1	1.0	1.0	1.0
8.25	1.6	1.5	1.4	1.4	1.3	1.3	1.2	1.2	1.1	1.1	1.1	1.0
8.50	1.6	1.6	1.5	1.4	1.4	1.3	1.3	1.2	1.2	1.1	1.1	1.0
8.75	1.7	1.6	1.5	1.5	1.4	1.3	1.3	1.2	1.2	1.1	1.1	1.1
9.00	1.7	1.6	1.5	1.5	1.4	1.4	1.3	1.3	1.2	1.2	1.2	1.1
9.25	1.7	1.7	1.6	1.5	1.5	1.4	1.4	1.3	1.3	1.2	1.2	1.1
9.50	1.8	1.7	1.6	1.6	1.5	1.5	1.4	1.4	1.3	1.3	1.2	1.2
9.75	1.8	1.8	1.7	1.6	1.6	1.5	1.5	1.4	1.4	1.3	1.3	1.2
10.00	1.9	1.8	1.7	1.7	1.6	1.6	1.5	1.4	1.4	1.4	1.3	1.3
10.25	1.9	1.9	1.8	1.7	1.7	1.6	1.5	1.5	1.4	1.4	1.4	1.3
10.50	2.0	1.9	1.8	1.8	1.7	1.7	1.6	1.5	1.5	1.4	1.4	1.4
10.75	2.0	2.0	1.9	1.8	1.8	1.7	1.6	1.6	1.5	1.5	1.5	1.4
11.00	2.1	2.0	1.9	1.8	1.8	1.7	1.7	1.6	1.6	1.5	1.5	1.4
11.25	2.1	2.0	1.9	1.9	1.8	1.8	1.7	1.6	1.6	1.5	1.5	1.5
11.50	2.1	2.1	2.0	1.9	1.9	1.8	1.7	1.7	1.6	1.6	1.6	1.5
11.75	2.2	2.1	2.0	1.9	1.9	1.8	1.8	1.7	1.7	1.6	1.6	1.5
12.00	2.2	2.2	2.1	2.0	1.9	1.9	1.8	1.8	1.7	1.7	1.7	1.6
12.25	2.3	2.2	2.1	2.0	2.0	1.9	1.9	1.8	1.8	1.7	1.7	1.6
12.50	2.3	2.3	2.2	2.1	2.0	2.0	1.9	1.9	1.8	1.8	1.8	1.7
12.75	2.3	2.3	2.2	2.1	2.1	2.0	2.0	1.9	1.9	1.8	1.8	1.7
13.00	2.4	2.3	2.3	2.2	2.1	2.1	2.0	2.0	1.9	1.9	1.9	1.8

* Yearly total of "monthly heat index"

TABLE A-2 (Continued)

	I											
T°C	25.0	27.5	30.0	32.5	35.0	37.5	40.0	42.5	45.0	47.5	50.0	52.5
13.25	2.4	2.4	2.3	2.2	2.2	2.1	2.1	2.0	2.0	1.9	1.9	1.8
13.50	2.5	2.4	2.3	2.3	2.2	2.2	2.1	2.1	2.0	2.0	2.0	1.9
13.75	2.5	2.4	2.4	2.3	2.3	2.2	2.1	2.1	2.0	2.0	2.0	1.9
14.00	2.5	2.5	2.4	2.3	2.3	2.2	2.2	2.2	2.1	2.0	2.0	1.9
14.25	2.6	2.5	2.4	2.4	2.3	2.3	2.2	2.2	2.1	2.1	2.1	2.0
14.50	2.6	2.5	2.5	2.4	2.4	2.3	2.3	2.3	2.2	2.1	2.1	2.0
14.75	2.7	2.6	2.5	2.5	2.4	2.4	2.3	2.3	2.2	2.2	2.2	2.1
15.00	2.7	2.6	2.6	2.5	2.5	2.4	2.4	2.3	2.3	2.2	2.2	2.1
15.25	2.8	2.7	2.6	2.6	2.5	2.5	2.4	2.4	2.3	2.3	2.3	2.2
15.50	2.8	2.7	2.7	2.6	2.6	2.5	2.5	2.4	2.4	2.3	2.3	2.2
15.75	2.9	2.8	2.7	2.7	2.6	2.6	2.5	2.5	2.4	2.4	2.4	2.3
16.00	2.9	2.8	2.8	2.7	2.7	2.6	2.6	2.5	2.5	2.4	2.4	2.3
16.25	2.9	2.9	2.8	2.7	2.7	2.6	2.6	2.5	2.5	2.4	2.4	2.4
16.50	3.0	2.9	2.8	2.8	2.7	2.7	2.6	2.5	2.5	2.5	2.5	2.4
16.75	3.0	2.9	2.9	2.8	2.8	2.7	2.7	2.6	2.5	2.5	2.5	2.5
17.00	3.0	3.0	2.9	2.9	2.8	2.8	2.7	2.7	2.6	2.6	2.6	2.5
17.25	3.1	3.0	2.9	2.9	2.9	2.8	2.8	2.7	2.7	2.6	2.6	2.6
17.50	3.1	3.0	3.0	3.0	2.9	2.9	2.8	2.8	2.7	2.7	2.7	2.6
17.75	3.2	3.1	3.0	3.0	3.0	2.9	2.9	2.8	2.8	2.8	2.7	2.7
18.00	3.2	3.1	3.1	3.1	3.0	3.0	2.9	2.9	2.9	2.8	2.8	2.7
18.25	3.2	3.2	3.1	3.1	3.1	3.0	3.0	2.9	2.9	2.9	2.8	2.8
18.50	3.3	3.2	3.2	3.1	3.1	3.1	3.0	3.0	3.0	2.9	2.9	2.8
18.75	3.3	3.2	3.2	3.2	3.1	3.1	3.0	3.0	3.0	2.9	2.9	2.9
19.00	3.3	3.3	3.2	3.2	3.2	3.1	3.1	3.0	3.0	3.0	3.0	2.9
19.25	3.4	3.3	3.3	3.2	3.2	3.2	3.1	3.1	3.1	3.0	3.0	3.0
19.50	3.4	3.4	3.3	3.3	3.3	3.2	3.2	3.1	3.1	3.1	3.1	3.0
19.75	3.5	3.4	3.4	3.3	3.3	3.3	3.2	3.2	3.2	3.1	3.1	3.1
20.00	3.5	3.5	3.4	3.4	3.4	3.3	3.3	3.2	3.2	3.2	3.2	3.1
20.25	3.6	3.5	3.5	3.4	3.4	3.4	3.3	3.3	3.3	3.2	3.2	3.2
20.50	3.6	3.5	3.5	3.5	3.5	3.4	3.4	3.3	3.3	3.3	3.3	3.2
20.75	3.7	3.6	3.5	3.5	3.5	3.5	3.4	3.4	3.4	3.3	3.3	3.3
21.00	3.7	3.6	3.6	3.5	3.5	3.5	3.5	3.4	3.4	3.4	3.4	3.3
21.25	3.7	3.6	3.6	3.6	3.6	3.5	3.5	3.4	3.4	3.4	3.4	3.4
21.50	3.8	3.7	3.6	3.6	3.6	3.6	3.5	3.5	3.5	3.5	3.5	3.4
21.75	3.8	3.7	3.7	3.7	3.6	3.6	3.6	3.5	3.5	3.5	3.5	3.5
22.00	3.8	3.8	3.7	3.7	3.7	3.7	3.6	3.6	3.6	3.6	3.6	3.5
22.25	3.9	3.9	3.8	3.8	3.7	3.7	3.7	3.7	3.6	3.6	3.6	3.6
22.50	3.9	3.8	3.8	3.8	3.8	3.8	3.7	3.7	3.7	3.7	3.7	3.6
22.75	4.0	3.9	3.9	3.8	3.8	3.8	3.8	3.8	2.7	2.7	2.7	2.7
23.00	4.0	3.9	3.9	3.9	3.9	3.9	3.8	3.8	3.8	3.8	3.8	3.8
23.25	4.1	4.0	4.0	3.9	3.9	3.9	3.9	3.9	3.8	3.8	3.8	3.8
23.50	4.1	4.0	4.0	4.0	4.0	4.0	3.9	3.9	3.9	3.9	3.9	3.9
23.75	4.1	4.0	4.0	4.0	4.0	4.0	4.0	3.9	3.9	3.9	3.9	3.9
24.00	4.2	4.1	4.1	4.0	4.0	4.0	4.0	4.0	4.0	4.0	4.0	3.9
24.25	4.2	4.1	4.1	4.1	4.1	4.1	4.1	4.0	4.0	4.0	4.0	4.0
24.50	4.2	4.1	4.1	4.1	4.1	4.1	4.1	4.1	4.1	4.1	4.1	4.1
24.75	4.3	4.2	4.2	4.2	4.2	4.2	4.2	4.1	4.1	4.1	4.1	4.1
25.00	4.3	4.2	4.2	4.2	4.2	4.2	4.2	4.2	4.2	4.2	4.2	4.2
25.25	4.3	4.3	4.3	4.3	4.3	4.3	4.3	4.2	4.2	4.2	4.2	4.2
25.50	4.4	4.3	4.3	4.3	4.3	4.3	4.3	4.3	4.3	4.3	4.3	4.3
25.75	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.4
26.00	4.5	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.4
26.25	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5
26.50	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5

TABLE A-2 ((Continued)

	I											
r_c	55.0	57.5	60.0	62.5	65.0	67.5	70.0	72.5	75.0	77.5	80.0	82.5
0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
.25	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
.50	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
.75	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.00	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.25	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.50	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.75	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2.00	0.1	0.1	0.1	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0
2.25	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.0
2.50	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
2.75	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
3.00	0.3	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
3.25	0.3	0.3	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1
3.50	0.3	0.3	0.3	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1
3.75	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1
4.00	0.4	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.1	0.1
4.25	0.4	0.4	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.1
4.50	0.4	0.4	0.4	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2
4.75	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2
5.00	0.5	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.3	0.3	0.2	0.2
5.25	0.5	0.5	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.3	0.2	0.2
5.50	0.5	0.5	0.5	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.3	0.3
5.75	0.6	0.5	0.5	0.5	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.3
6.00	0.6	0.6	0.5	0.5	0.5	0.4	0.4	0.4	0.4	0.3	0.3	0.3
6.25	0.6	0.6	0.6	0.5	0.5	0.4	0.4	0.4	0.4	0.4	0.3	0.3
6.50	0.7	0.6	0.6	0.6	0.5	0.5	0.4	0.4	0.4	0.4	0.4	0.4
6.75	0.7	0.7	0.6	0.6	0.6	0.5	0.4	0.4	0.4	0.4	0.4	0.4
7.00	0.7	0.7	0.7	0.6	0.6	0.5	0.5	0.5	0.5	0.4	0.4	0.4
7.25	0.8	0.7	0.7	0.7	0.6	0.6	0.5	0.5	0.5	0.5	0.5	0.4
7.50	0.8	0.8	0.7	0.7	0.7	0.6	0.6	0.5	0.5	0.5	0.5	0.4
7.75	0.9	0.8	0.8	0.7	0.7	0.6	0.6	0.6	0.6	0.5	0.5	0.5
8.00	0.9	0.9	0.8	0.8	0.7	0.7	0.6	0.6	0.6	0.6	0.5	0.5
8.25	0.9	0.9	0.8	0.8	0.8	0.7	0.7	0.6	0.6	0.6	0.6	0.5
8.50	1.0	0.9	0.9	0.8	0.8	0.8	0.7	0.7	0.7	0.6	0.6	0.6
8.75	1.0	1.0	0.9	0.9	0.8	0.8	0.7	0.7	0.7	0.7	0.6	0.6
9.00	1.0	1.0	0.9	0.9	0.9	0.8	0.8	0.7	0.7	0.7	0.7	0.6
9.25	1.1	1.0	1.0	0.9	0.9	0.9	0.8	0.8	0.8	0.7	0.7	0.7
9.50	1.1	1.1	1.0	1.0	0.9	0.9	0.9	0.8	0.8	0.8	0.7	0.7
9.75	1.2	1.1	1.1	1.0	1.0	0.9	0.9	0.8	0.8	0.8	0.8	0.7
10.00	1.2	1.2	1.1	1.1	1.0	1.0	0.9	0.9	0.9	0.8	0.8	0.8
10.25	1.3	1.2	1.2	1.1	1.1	1.0	1.0	0.9	0.9	0.9	0.8	0.8
10.50	1.3	1.2	1.2	1.2	1.1	1.1	1.0	1.0	1.0	0.9	0.9	0.8
10.75	1.3	1.3	1.2	1.2	1.2	1.1	1.1	1.0	1.0	0.9	0.9	0.9
11.00	1.4	1.3	1.3	1.3	1.2	1.1	1.1	1.0	1.0	1.0	1.0	0.9
11.25	1.4	1.3	1.3	1.3	1.2	1.2	1.1	1.1	1.1	1.0	1.0	0.9
11.50	1.4	1.4	1.3	1.3	1.3	1.2	1.2	1.1	1.1	1.0	1.0	1.0
11.75	1.5	1.4	1.4	1.4	1.3	1.3	1.2	1.1	1.1	1.1	1.1	1.0
12.00	1.5	1.5	1.4	1.4	1.4	1.3	1.3	1.2	1.2	1.1	1.1	1.0
12.25	1.6	1.5	1.5	1.5	1.4	1.4	1.3	1.2	1.2	1.2	1.2	1.1
12.50	1.6	1.6	1.5	1.5	1.5	1.4	1.4	1.3	1.3	1.2	1.2	1.2
12.75	1.7	1.6	1.6	1.5	1.5	1.5	1.4	1.4	1.3	1.3	1.2	1.2
13.00	1.7	1.7	1.6	1.6	1.5	1.5	1.5	1.5	1.3	1.3	1.3	1.2

TABLE A-2 (Continued)

	I											
T°C	55.0	57.5	60.0	62.5	65.0	67.5	70.0	72.5	75.0	77.5	80.0	82.5
13.25	1.8	1.7	1.7	1.6	1.6	1.6	1.5	1.5	1.4	1.4	1.3	1.3
13.50	1.8	1.8	1.7	1.7	1.6	1.6	1.6	1.5	1.4	1.4	1.4	1.3
13.75	1.9	1.8	1.8	1.7	1.7	1.7	1.6	1.6	1.5	1.5	1.4	1.4
14.00	1.9	1.8	1.8	1.7	1.7	1.7	1.6	1.6	1.5	1.5	1.5	1.4
14.25	2.0	1.9	1.9	1.8	1.7	1.7	1.7	1.7	1.6	1.5	1.5	1.5
14.50	2.0	1.9	1.9	1.8	1.8	1.8	1.7	1.7	1.6	1.6	1.6	1.5
14.75	2.1	2.0	2.0	1.9	1.8	1.8	1.8	1.7	1.7	1.6	1.6	1.6
15.00	2.1	2.0	2.0	1.9	1.9	1.9	1.8	1.8	1.7	1.7	1.7	1.6
15.25	2.2	2.1	2.1	2.0	1.9	1.9	1.9	1.8	1.8	1.7	1.7	1.7
15.50	2.2	2.1	2.1	2.0	2.0	1.9	1.9	1.9	1.8	1.8	1.8	1.7
15.75	2.3	2.2	2.2	2.1	2.0	2.0	2.0	1.9	1.9	1.8	1.8	1.8
16.00	2.3	2.2	2.2	2.1	2.1	2.0	2.0	2.0	2.0	1.9	1.9	1.8
16.25	2.3	2.2	2.2	2.1	2.1	2.0	2.0	2.0	2.0	1.9	1.9	1.9
16.50	2.4	2.3	2.3	2.2	2.1	2.1	2.1	2.0	2.0	1.9	1.9	1.9
16.75	2.4	2.3	2.3	2.2	2.2	2.1	2.1	2.1	2.1	2.0	2.0	2.0
17.00	2.5	2.4	2.4	2.3	2.3	2.2	2.2	2.1	2.1	2.0	2.0	2.0
17.25	2.5	2.5	2.4	2.4	2.3	2.3	2.2	2.2	2.2	2.1	2.1	2.1
17.50	2.6	2.5	2.5	2.4	2.4	2.3	2.3	2.3	2.2	2.2	2.2	2.1
17.75	2.6	2.6	2.5	2.5	2.4	2.4	2.4	2.3	2.3	2.3	2.2	2.2
18.00	2.7	2.6	2.6	2.5	2.5	2.4	2.4	2.4	2.3	2.3	2.3	2.3
18.25	2.7	2.7	2.6	2.5	2.5	2.5	2.5	2.4	2.4	2.4	2.4	2.4
18.50	2.8	2.7	2.7	2.6	2.5	2.5	2.5	2.5	2.4	2.4	2.4	2.4
18.75	2.8	2.8	2.7	2.7	2.6	2.6	2.6	2.5	2.5	2.5	2.5	2.5
19.00	2.9	2.8	2.8	2.7	2.7	2.6	2.6	2.5	2.5	2.5	2.5	2.5
19.25	2.9	2.9	2.9	2.8	2.7	2.7	2.7	2.6	2.6	2.6	2.6	2.5
19.50	3.0	2.9	2.9	2.8	2.8	2.7	2.7	2.7	2.7	2.6	2.6	2.6
19.75	3.0	3.0	2.9	2.9	2.9	2.8	2.8	2.7	2.7	2.7	2.7	2.7
20.00	3.1	3.0	3.0	3.0	2.9	2.9	2.8	2.8	2.8	2.8	2.8	2.8
20.25	3.2	3.1	3.1	3.0	3.0	3.0	2.9	2.9	2.9	2.8	2.8	2.8
20.50	3.2	3.2	3.2	3.1	3.1	3.0	3.0	3.0	2.9	2.9	2.9	2.9
20.75	3.3	3.2	3.2	3.1	3.1	3.1	3.0	3.0	3.0	3.0	3.0	3.0
21.00	3.3	3.3	3.3	3.2	3.2	3.1	3.1	3.1	3.0	3.0	3.0	3.0
21.25	3.4	3.3	3.3	3.2	3.2	3.2	3.1	3.1	3.1	3.1	3.1	3.1
21.50	3.4	3.4	3.4	3.3	3.3	3.2	3.2	3.2	3.2	3.1	3.1	3.1
21.75	3.5	3.4	3.4	3.3	3.3	3.3	3.2	3.2	3.2	3.2	3.2	3.2
22.00	3.5	3.5	3.4	3.4	3.4	3.3	3.3	3.3	3.3	3.3	3.3	3.3
22.25	3.6	3.5	3.5	3.4	3.4	3.4	3.3	3.3	3.3	3.3	3.3	3.3
22.50	3.6	3.6	3.6	3.5	3.5	3.5	3.4	3.4	3.4	3.4	3.4	3.4
22.75	3.7	3.7	3.6	3.6	3.6	3.6	3.5	3.5	3.5	3.5	3.5	3.5
23.00	3.7	3.7	3.7	3.6	3.6	3.6	3.6	3.6	3.6	3.6	3.6	3.6
23.25	3.8	3.8	3.8	3.7	3.7	3.7	3.7	3.7	3.7	3.7	3.7	3.7
23.50	3.8	3.8	3.8	3.8	3.8	3.8	3.8	3.7	3.7	3.7	3.7	3.7
23.75	3.9	3.9	3.9	3.9	3.8	3.8	3.8	3.8	3.8	3.8	3.8	3.8
24.00	3.9	3.9	3.9	3.9	3.9	3.9	3.9	3.8	3.8	3.8	3.8	3.8
24.25	4.0	4.0	4.0	3.9	3.9	3.9	3.9	3.9	3.9	3.9	3.9	3.9
24.50	4.0	4.0	4.0	4.0	4.0	4.0	4.0	3.9	3.9	3.9	3.9	3.9
24.75	4.1	4.1	4.0	4.0	4.0	4.0	4.0	4.0	4.0	4.0	4.0	4.0
25.00	4.2	4.1	4.1	4.1	4.1	4.1	4.1	4.1	4.1	4.0	4.0	4.0
25.25	4.2	4.2	4.2	4.2	4.2	4.2	4.1	4.1	4.1	4.1	4.1	4.1
25.50	4.3	4.3	4.3	4.3	4.3	4.2	4.2	4.2	4.2	4.2	4.2	4.2
25.75	4.3	4.3	4.3	4.3	4.3	4.3	4.3	4.3	4.3	4.3	4.2	4.2
26.00	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.3	4.3
26.25	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.4	4.4
26.50	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5

TABLE A-2 (Continued)

	I											
T°C	85.0	87.5	90.0	92.5	95.0	97.5	100.0	102.5	105.0	107.5	110.0	112.5
0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
.25	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
.50	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
.75	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.00	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.25	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.50	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.75	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2.00	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2.25	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2.50	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2.75	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3.00	0.1	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3.25	0.1	0.1	0.1	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0
3.50	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0
3.75	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0
4.00	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0
4.25	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0
4.50	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
4.75	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
5.00	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
5.25	0.2	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1
5.50	0.2	0.2	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1
5.75	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1
6.00	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1
6.25	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1
6.50	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1
6.75	0.3	0.3	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2	-0.2	0.1
7.00	0.4	0.3	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2
7.25	0.4	0.4	0.3	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2
7.50	0.4	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2
7.75	0.5	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.3	0.2	0.2	0.2
8.00	0.5	0.5	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.3	0.3	0.2
8.25	0.5	0.5	0.5	0.4	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.3
8.50	0.6	0.5	0.5	0.5	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.3
8.75	0.6	0.5	0.5	0.5	0.4	0.4	0.4	0.4	0.4	0.3	0.3	0.3
9.00	0.6	0.6	0.5	0.5	0.5	0.4	0.4	0.4	0.4	0.4	0.3	0.3
9.25	0.6	0.6	0.6	0.5	0.5	0.5	0.5	0.4	0.4	0.4	0.4	0.3
9.50	0.7	0.6	0.6	0.6	0.5	0.5	0.5	0.4	0.4	0.4	0.4	0.4
9.75	0.7	0.7	0.6	0.6	0.6	0.5	0.5	0.5	0.5	0.4	0.4	0.4
10.00	0.7	0.7	0.7	0.6	0.6	0.6	0.5	0.5	0.5	0.4	0.4	0.4
10.25	0.8	0.7	0.7	0.7	0.6	0.6	0.6	0.5	0.5	0.5	0.5	0.4
10.50	0.8	0.8	0.7	0.7	0.7	0.7	0.6	0.6	0.5	0.5	0.5	0.5
10.75	0.8	0.8	0.8	0.8	0.7	0.7	0.6	0.6	0.6	0.5	0.5	0.5
11.00	0.9	0.8	0.8	0.8	0.7	0.7	0.7	0.6	0.6	0.6	0.6	0.5
11.25	0.9	0.9	0.8	0.8	0.8	0.7	0.7	0.6	0.6	0.6	0.6	0.5
11.50	0.9	0.9	0.9	0.9	0.8	0.8	0.7	0.7	0.7	0.6	0.6	0.6
11.75	1.0	0.9	0.9	0.9	0.8	0.8	0.8	0.7	0.7	0.7	0.6	0.6
12.00	1.0	1.0	0.9	0.9	0.9	0.8	0.8	0.7	0.7	0.7	0.7	0.6
12.25	1.1	1.0	1.0	1.0	0.9	0.9	0.8	0.8	0.8	0.8	0.7	0.7
12.50	1.1	1.1	1.0	1.0	0.9	0.9	0.9	0.8	0.8	0.8	0.7	0.7
12.75	1.2	1.1	1.1	1.1	1.0	1.0	0.9	0.9	0.8	0.8	0.8	0.7
13.00	1.2	1.2	1.1	1.1	1.0	1.0	1.0	0.9	0.9	0.9	0.8	0.8

TABLE A-2 (Continued)

T°C	I 85.0	87.5	90.0	92.5	95.0	97.5	100.0	102.5	105.0	107.5	110.0	112.5
13.25	1.2	1.2	1.2	1.1	1.1	1.0	1.0	1.0	0.9	0.9	0.9	0.8
13.50	1.3	1.2	1.2	1.2	1.1	1.1	1.0	1.0	1.0	0.9	0.9	0.9
13.75	1.3	1.3	1.2	1.2	1.1	1.1	1.1	1.1	1.0	1.0	0.9	0.9
14.00	1.4	1.3	1.3	1.2	1.2	1.1	1.1	1.1	1.0	1.0	1.0	0.9
14.25	1.4	1.3	1.3	1.3	1.2	1.2	1.2	1.1	1.1	1.0	1.0	1.0
14.50	1.5	1.4	1.4	1.3	1.3	1.2	1.2	1.2	1.1	1.1	1.0	1.0
14.75	1.5	1.4	1.4	1.3	1.3	1.3	1.3	1.2	1.1	1.1	1.1	1.1
15.00	1.6	1.5	1.5	1.4	1.4	1.3	1.3	1.3	1.2	1.1	1.1	1.1
15.25	1.6	1.6	1.5	1.5	1.5	1.4	1.4	1.3	1.2	1.2	1.2	1.2
15.50	1.7	1.6	1.6	1.6	1.5	1.5	1.4	1.4	1.3	1.3	1.3	1.2
15.75	1.7	1.7	1.7	1.6	1.6	1.5	1.5	1.4	1.3	1.3	1.3	1.3
16.00	1.8	1.7	1.7	1.7	1.6	1.6	1.5	1.5	1.4	1.4	1.4	1.3
16.25	1.8	1.8	1.8	1.7	1.7	1.6	1.6	1.5	1.4	1.4	1.4	1.4
16.50	1.9	1.9	1.8	1.7	1.7	1.7	1.6	1.6	1.5	1.5	1.5	1.4
16.75	1.9	1.9	1.9	1.8	1.8	1.7	1.7	1.6	1.5	1.5	1.5	1.5
17.00	2.0	1.9	1.9	1.8	1.8	1.8	1.7	1.7	1.7	1.6	1.6	1.5
17.25	2.1	2.0	2.0	1.9	1.9	1.8	1.8	1.7	1.7	1.6	1.6	1.5
17.50	2.1	2.0	2.0	2.0	1.9	1.9	1.8	1.8	1.7	1.7	1.7	1.6
17.75	2.2	2.1	2.1	2.0	2.0	1.9	1.9	1.8	1.8	1.8	1.8	1.7
18.00	2.3	2.2	2.1	2.1	2.1	2.0	2.0	1.9	1.9	1.8	1.8	1.8
18.25	2.3	2.2	2.2	2.2	2.2	2.1	2.1	2.0	1.9	1.9	1.9	1.8
18.50	2.4	2.3	2.2	2.2	2.2	2.2	2.2	2.0	2.0	1.9	1.9	1.9
18.75	2.4	2.3	2.3	2.3	2.3	2.2	2.2	2.1	2.0	2.0	2.0	2.0
19.00	2.5	2.4	2.4	2.3	2.3	2.3	2.3	2.1	2.1	2.0	2.0	2.0
19.25	2.5	2.4	2.4	2.4	2.4	2.3	2.3	2.2	2.1	2.1	2.1	2.1
19.50	2.6	2.5	2.5	2.4	2.4	2.4	2.4	2.2	2.2	2.2	2.2	2.1
19.75	2.6	2.5	2.5	2.5	2.5	2.4	2.4	2.3	2.2	2.2	2.2	2.2
20.00	2.7	2.6	2.6	2.6	2.5	2.5	2.5	2.4	2.3	2.3	2.3	2.3
20.25	2.8	2.7	2.7	2.6	2.6	2.6	2.6	2.5	2.4	2.4	2.4	2.4
20.50	2.8	2.8	2.8	2.7	2.7	2.7	2.6	2.5	2.5	2.5	2.5	2.5
20.75	2.9	2.8	2.8	2.8	2.8	2.7	2.7	2.6	2.6	2.6	2.6	2.6
21.00	3.0	2.9	2.9	2.9	2.8	2.8	2.8	2.7	2.7	2.7	2.7	2.6
21.25	3.0	2.9	2.9	2.9	2.9	2.9	2.8	2.8	2.7	2.7	2.7	2.7
21.50	3.1	3.0	3.0	2.9	2.9	2.9	2.9	2.8	2.8	2.8	2.8	2.7
21.75	3.2	3.1	3.1	3.0	3.0	3.0	3.0	2.9	2.9	2.9	2.9	2.8
22.00	3.2	3.1	3.1	3.1	3.1	3.1	3.1	3.0	3.0	3.0	3.0	2.9
22.25	3.3	3.2	3.2	3.2	3.2	3.2	3.2	3.1	3.0	3.0	3.0	3.0
22.50	3.4	3.3	3.3	3.2	3.2	3.2	3.2	3.1	3.1	3.1	3.1	3.1
22.75	3.5	3.4	3.4	3.3	3.3	3.3	3.3	3.2	3.2	3.2	3.2	3.2
23.00	3.6	3.5	3.5	3.4	3.4	3.4	3.4	3.3	3.3	3.3	3.3	3.3
23.25	3.6	3.5	3.5	3.5	3.5	3.5	3.5	3.4	3.3	3.3	3.3	3.3
23.50	3.7	3.6	3.6	3.6	3.6	3.5	3.5	3.5	3.4	3.4	3.4	3.4
23.75	3.7	3.7	3.7	3.6	3.6	3.6	3.6	3.6	3.5	3.5	3.5	3.4
24.00	3.8	3.7	3.7	3.7	3.7	3.7	3.6	3.6	3.5	3.5	3.5	3.5
24.25	3.9	3.8	3.8	3.8	3.8	3.8	3.7	3.7	3.6	3.6	3.6	3.6
24.50	3.9	3.8	3.8	3.8	3.8	3.8	3.8	3.8	3.7	3.7	3.7	3.7
24.75	4.0	3.9	3.9	3.9	3.9	3.9	3.9	3.9	3.8	3.8	3.8	3.8
25.00	4.0	4.0	4.0	4.0	4.0	4.0	4.0	4.0	3.9	3.9	3.9	3.9
25.25	4.1	4.1	4.1	4.1	4.1	4.1	4.1	4.1	4.0	4.0	4.0	4.0
25.50	4.2	4.2	4.2	4.2	4.2	4.2	4.2	4.2	4.1	4.1	4.1	4.1
25.75	4.3	4.3	4.3	4.3	4.3	4.3	4.3	4.3	4.2	4.2	4.2	4.2
26.00	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.3	4.3	4.3	4.3
26.25	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.4	4.4	4.4	4.4
26.50	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5

TABLE. A-2 (Continued)

	I										
T°C	115.0	117.5	120.0	122.5	125.0	127.5	130.0	132.5	135.0	137.5	140.0
0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
.25	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
.50	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
.75	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.00	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.25	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.50	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.75	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2.00	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2.25	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2.50	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2.75	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3.00	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3.25	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3.50	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3.75	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4.00	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4.25	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4.50	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4.75	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5.00	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5.25	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5.50	0.1	0.1	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0
5.75	0.1	0.1	0.1	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0
6.00	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.0	0.0	0.0	0.0
6.25	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.0	0.0	0.0
6.50	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.0	0.0	0.0
6.75	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.0
7.00	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.0
7.25	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
7.50	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
7.75	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
8.00	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1
8.25	0.2	0.2	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.1
8.50	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.1
8.75	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.1	0.1	0.1
9.00	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.1	0.1	0.1
9.25	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.1	0.1	0.1
9.50	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.1
9.75	0.4	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2
10.00	0.4	0.3	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2
10.25	0.4	0.4	0.3	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2
10.50	0.4	0.4	0.4	0.3	0.3	0.3	0.3	0.3	0.2	0.2	0.2
10.75	0.5	0.4	0.4	0.4	0.4	0.3	0.3	0.3	0.2	0.2	0.2
11.00	0.5	0.5	0.4	0.4	0.4	0.4	0.3	0.3	0.3	0.2	0.2
11.25	0.5	0.5	0.4	0.4	0.4	0.4	0.4	0.3	0.3	0.3	0.3
11.50	0.5	0.5	0.5	0.4	0.4	0.4	0.4	0.3	0.3	0.3	0.3
11.75	0.6	0.5	0.5	0.4	0.4	0.4	0.4	0.4	0.3	0.3	0.3
12.00	0.6	0.5	0.5	0.5	0.5	0.4	0.4	0.4	0.3	0.3	0.3
12.25	0.5	0.6	0.6	0.5	0.5	0.5	0.5	0.4	0.4	0.3	0.3
12.50	0.7	0.6	0.6	0.5	0.5	0.5	0.5	0.5	0.4	0.4	0.4
12.75	0.7	0.6	0.5	0.6	0.6	0.6	0.5	0.5	0.4	0.4	0.4
13.00	0.7	0.7	0.7	0.6	0.6	0.6	0.6	0.5	0.5	0.4	0.4

TABLE A-2 (Concluded)

T°C	I 115.0	117.5	120.0	122.5	125.0	127.5	130.0	132.5	135.0	137.5	140.0
13.25	0.7	0.7	0.7	0.6	0.6	0.6	0.6	0.6	0.5	0.5	0.5
13.50	0.8	0.8	0.7	0.7	0.7	0.6	0.6	0.6	0.5	0.5	0.5
13.75	0.8	0.8	0.8	0.7	0.7	0.7	0.6	0.5	0.6	0.5	0.5
14.00	0.8	0.8	0.8	0.7	0.7	0.7	0.7	0.6	0.6	0.5	0.5
14.25	0.9	0.9	0.8	0.8	0.8	0.7	0.7	0.7	0.6	0.5	0.5
14.50	0.9	0.9	0.9	0.8	0.8	0.8	0.7	0.7	0.6	0.6	0.6
14.75	1.0	1.0	0.9	0.9	0.8	0.8	0.8	0.7	0.7	0.6	0.6
15.00	1.0	1.0	1.0	0.9	0.9	0.9	0.8	0.8	0.7	0.7	0.7
15.25	1.1	1.1	1.0	1.0	0.9	0.9	0.9	0.8	0.8	0.7	0.7
15.50	1.2	1.1	1.1	1.0	1.0	1.0	0.9	0.9	0.8	0.8	0.8
15.75	1.2	1.1	1.1	1.1	1.1	1.0	1.0	0.9	0.9	0.8	0.8
16.00	1.3	1.2	1.2	1.1	1.1	1.1	1.0	1.0	0.9	0.9	0.9
16.25	1.3	1.2	1.2	1.2	1.1	1.1	1.1	1.0	1.0	0.9	0.9
16.50	1.4	1.3	1.2	1.2	1.2	1.1	1.1	1.1	1.0	0.9	0.9
16.75	1.4	1.3	1.3	1.2	1.2	1.2	1.2	1.1	1.0	1.0	1.0
17.00	1.5	1.4	1.3	1.3	1.3	1.3	1.2	1.2	1.1	1.0	1.0
17.25	1.6	1.5	1.4	1.3	1.3	1.3	1.3	1.2	1.2	1.1	1.1
17.50	1.6	1.5	1.5	1.4	1.4	1.4	1.3	1.3	1.2	1.2	1.2
17.75	1.7	1.6	1.5	1.5	1.5	1.4	1.4	1.4	1.3	1.2	1.2
18.00	1.8	1.7	1.6	1.5	1.5	1.5	1.5	1.5	1.4	1.3	1.3
18.25	1.8	1.7	1.6	1.6	1.6	1.6	1.5	1.5	1.4	1.4	1.4
18.50	1.9	1.8	1.7	1.6	1.6	1.6	1.6	1.6	1.5	1.4	1.4
18.75	1.9	1.8	1.8	1.7	1.7	1.7	1.6	1.6	1.5	1.5	1.5
19.00	1.9	1.8	1.8	1.7	1.7	1.7	1.7	1.7	1.6	1.5	1.5
19.25	2.0	1.9	1.9	1.8	1.8	1.8	1.7	1.7	1.7	1.6	1.6
19.50	2.1	2.0	1.9	1.9	1.9	1.9	1.8	1.8	1.7	1.7	1.6
19.75	2.2	2.0	2.0	1.9	1.9	1.9	1.9	1.9	1.8	1.8	1.7
20.00	2.2	2.1	2.1	2.0	2.0	2.0	2.0	2.0	1.9	1.9	1.8
20.25	2.3	2.2	2.2	2.1	2.1	2.1	2.1	2.1	2.0	2.0	1.9
20.50	2.4	2.3	2.3	2.2	2.2	2.2	2.2	2.2	2.1	2.1	2.0
20.75	2.5	2.4	2.3	2.3	2.3	2.3	2.3	2.2	2.2	2.1	2.1
21.00	2.6	2.5	2.4	2.4	2.3	2.3	2.3	2.3	2.2	2.2	2.2
21.25	2.7	2.6	2.5	2.4	2.4	2.4	2.4	2.3	2.3	2.3	2.3
21.50	2.7	2.6	2.5	2.5	2.4	2.4	2.4	2.4	2.3	2.3	2.3
21.75	2.8	2.7	2.6	2.5	2.5	2.5	2.5	2.5	2.4	2.4	2.4
22.00	2.9	2.8	2.7	2.6	2.6	2.6	2.6	2.5	2.6	2.5	2.5
22.25	3.0	2.9	2.8	2.7	2.7	2.7	2.7	2.7	2.7	2.6	2.6
22.50	3.1	3.0	2.9	2.8	2.8	2.8	2.8	2.8	2.8	2.7	2.7
22.75	3.2	3.1	3.0	2.9	2.9	2.9	2.9	2.9	2.9	2.8	2.8
23.00	3.3	3.2	3.1	3.0	3.0	3.0	3.0	3.0	3.0	2.9	2.9
23.25	3.3	3.3	3.2	3.1	3.1	3.1	3.1	3.1	3.1	3.0	3.0
23.50	3.4	3.4	3.3	3.2	3.2	3.2	3.2	3.2	3.2	3.1	3.1
23.75	3.5	3.5	3.4	3.3	3.2	3.2	3.2	3.2	3.2	3.2	3.1
24.00	3.5	3.5	3.4	3.3	3.3	3.3	3.3	3.3	3.3	3.3	3.2
24.25	3.6	3.6	3.5	3.4	3.4	3.4	3.4	3.4	3.4	3.4	3.3
24.50	3.7	3.7	3.6	3.5	3.5	3.5	3.5	3.5	3.5	3.5	3.4
24.75	3.8	3.8	3.7	3.6	3.6	3.6	3.6	3.6	3.6	3.6	3.5
25.00	3.9	3.9	3.8	3.8	3.8	3.8	3.8	3.8	3.8	3.8	3.7
25.25	4.0	4.0	3.9	3.9	3.9	3.9	3.9	3.9	3.9	3.9	3.8
25.50	4.1	4.1	4.0	4.0	4.0	4.0	4.0	4.0	4.0	4.0	3.9
25.75	4.2	4.2	4.2	4.2	4.2	4.2	4.2	4.2	4.2	4.2	4.1
26.00	4.3	4.3	4.3	4.3	4.3	4.3	4.3	4.3	4.3	4.3	4.2
26.25	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.3
26.50	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.4

TABLE A-3

VALUES OF UNADJUSTED DAILY POTENTIAL EVAPOTRANSPIRATION FOR MEAN TEMPERATURES ABOVE 26.5°C

UNADJUSTED POTENTIAL EVAPOTRANSPIRATION IN mm

T°C	0.0	0.1	0.2	0.3	0.4	0.5	0.6	0.7	0.8	0.9
26						4.5	4.5	4.6	4.6	4.6
27	4.6	4.7	4.7	4.7	4.8	4.8	4.8	4.8	4.9	4.9
28	4.9	5.0	5.0	5.0	5.0	5.1	5.1	5.1	5.1	5.2
29	5.2	5.2	5.2	5.2	5.3	5.3	5.3	5.3	5.4	5.4
30	5.4	5.4	5.4	5.5	5.5	5.5	5.5	5.5	5.6	5.6
31	5.6	5.6	5.6	5.6	5.7	5.7	5.7	5.7	5.7	5.8
32	5.8	5.8	5.8	5.8	5.8	5.8	5.9	5.9	5.9	5.9
33	5.9	5.9	5.8	5.9	6.0	6.0	6.0	6.0	6.0	6.0
34	6.0	6.0	6.0	6.0	6.1	6.1	6.1	6.1	6.1	6.1
35	6.1	6.1	6.1	6.1	6.1	6.1	6.1	6.1	5.1	6.1
36	6.1	6.1	6.2	6.2	6.2	6.2	6.2	6.2	6.2	6.2
37	6.2	6.2	6.2	6.2	6.2	6.2	6.2	6.2	6.2	6.2
38	6.2									

TABLE A-4

MEAN POSSIBLE MONTHLY DURATION OF SUNLIGHT IN THE NORTHERN HEMISPHERE
EXPRESSED IN UNITS OF 12 HOURS

NORTHERN LATITUDES	J	F	M	A	M	J	J	A	S	O	N	D
0°	31.2	28.2	31.2	30.3	31.2	30.3	31.2	31.2	30.3	31.2	30.3	31.2
1	31.2	28.2	31.2	30.3	31.2	30.3	31.2	31.2	30.3	31.2	30.3	31.2
2	31.2	28.2	31.2	30.3	31.5	30.6	31.2	31.2	30.3	31.2	30.0	30.9
3	30.9	28.2	30.9	30.3	31.5	30.6	31.5	31.2	30.3	31.2	30.0	30.9
4	30.9	27.9	30.9	30.6	31.8	30.9	31.5	31.5	30.3	30.9	30.0	30.6
5	30.6	27.9	30.9	30.6	31.8	30.9	31.8	31.5	30.3	30.9	29.7	30.6
6	30.6	27.9	30.9	30.6	31.8	31.2	31.8	31.5	30.3	30.9	29.7	30.3
7	30.3	27.6	30.9	30.6	32.1	31.2	32.1	31.8	30.3	30.9	29.7	30.3
8	30.3	27.6	30.9	30.9	32.1	31.5	32.1	31.8	30.6	30.6	29.4	30.0
9	30.0	27.6	30.9	30.9	32.4	31.5	32.4	31.8	30.6	30.6	29.4	30.0
10	30.0	27.3	30.9	30.9	32.4	31.8	32.4	32.1	30.6	30.6	29.4	29.7
11	29.7	27.3	30.9	30.9	32.7	31.8	32.7	32.1	30.6	30.6	29.1	29.7
12	29.7	27.3	30.9	31.2	32.7	32.1	33.0	32.1	30.6	30.3	29.1	29.4
13	29.4	27.3	30.9	31.2	33.0	32.1	33.0	32.4	30.6	30.3	28.8	29.4
14	29.4	27.3	30.9	31.2	33.0	32.4	33.3	32.4	30.6	30.3	28.8	29.1
15	29.1	27.3	30.9	31.2	33.3	32.4	33.6	32.4	30.6	30.3	28.5	29.1
16	29.1	27.3	30.9	31.2	33.3	32.7	33.6	32.7	30.6	30.3	28.5	28.8
17	28.8	27.3	30.9	31.5	33.6	32.7	33.9	32.7	30.6	30.0	28.2	28.5
18	28.8	27.0	30.9	31.5	33.6	33.0	33.9	33.0	30.6	30.0	28.2	28.5
19	28.5	27.0	30.9	31.5	33.9	33.0	34.2	33.0	30.6	30.0	27.9	28.5
20	28.5	27.0	30.9	31.5	33.9	33.3	34.2	33.3	30.6	30.0	27.9	28.2
21	28.2	27.0	30.9	31.5	33.9	33.3	34.5	33.3	30.6	30.0	27.6	28.2
22	28.2	26.7	30.9	31.8	34.2	33.6	34.5	33.3	30.6	29.7	27.6	27.9
23	27.9	26.7	30.9	31.8	34.2	33.9	34.8	33.6	30.6	29.7	27.6	27.6
24	27.9	26.7	30.9	31.8	34.5	34.2	34.8	33.6	30.6	29.7	27.3	27.6
25	27.9	26.7	30.9	31.8	34.5	34.2	35.1	33.6	30.6	29.7	27.3	27.3
26	27.6	26.4	30.9	32.1	34.8	34.5	35.1	33.6	30.6	29.7	27.3	27.3
27	27.6	26.4	30.9	32.1	34.8	34.5	35.4	33.9	30.6	29.7	27.0	27.0
28	27.3	26.4	30.9	32.1	35.1	34.8	35.4	33.9	30.9	29.4	27.0	27.0
29	27.3	26.1	30.9	32.1	35.1	34.8	35.7	33.9	30.9	29.4	26.7	26.7
30	27.0	26.1	30.9	32.4	35.4	35.1	36.0	34.2	30.9	29.4	26.7	26.4
31	27.0	26.1	30.9	32.4	35.4	35.1	36.0	34.2	30.9	29.4	26.4	26.4
32	26.7	25.8	30.9	32.4	35.7	35.4	36.3	34.5	30.9	29.4	26.4	26.1
33	26.4	25.8	30.9	32.7	35.7	35.7	36.3	34.5	30.9	29.1	26.1	25.8
34	26.4	25.8	30.9	32.7	36.0	36.0	36.6	34.8	30.9	29.1	26.1	25.8
35	26.1	25.5	30.9	32.7	36.3	36.3	36.9	34.8	30.9	29.1	25.8	25.5
36	26.1	25.5	30.9	33.0	36.3	36.6	37.2	34.8	30.9	29.1	25.5	25.2
37	25.8	25.5	30.9	33.0	[illegible]	36.9	37.5	35.1	30.9	29.1	25.5	24.9
38	25.5	25.2	30.9	33.0	[illegible]	37.2	37.5	35.1	31.2	28.8	25.2	24.9
39	25.5	25.2	30.9	33.3	36.9	37.2	37.8	35.4	31.2	28.8	25.2	24.6
40	25.2	24.9	30.9	33.3	37.2	37.5	38.1	35.4	31.2	28.8	24.9	24.3
41	24.9	24.9	30.9	33.3	37.5	37.8	38.1	35.7	31.2	28.8	24.6	24.0
42	24.6	24.6	30.9	33.6	37.8	38.1	38.4	35.7	31.2	28.5	24.6	23.7
43	24.3	24.6	30.6	33.6	37.8	38.4	38.7	36.0	31.2	28.5	24.3	23.1
44	24.3	24.3	30.6	33.6	38.1	38.7	39.0	36.0	31.2	28.5	24.0	22.8
45	24.0	24.3	30.6	33.9	38.4	38.7	39.3	36.3	31.2	28.2	23.7	22.5
46	23.7	24.0	30.6	33.9	38.7	39.0	39.6	36.6	31.2	28.2	23.7	22.2
47	23.1	24.0	30.6	34.2	39.0	39.6	39.9	36.6	31.5	27.9	23.4	21.9
48	22.8	23.7	30.6	34.2	39.3	39.9	40.2	36.9	31.5	27.9	23.1	21.6
49	22.5	23.7	30.6	34.5	39.6	40.2	40.5	37.2	31.5	27.6	22.8	21.3
50	22.2	23.4	30.6	34.5	39.9	40.8	41.1	37.5	31.8	27.6	22.8	21.0

TABLE A-5

PROVISIONAL WATER HOLDING CAPACITIES WITH DIFFERENT COMBINATIONS OF SOIL AND VEGETATION

SOIL TYPE	AVAILABLE WATER		ROOT ZONE		APPLICABLE SOIL MOISTURE RETENTION TABLE	
	MM/M	IN/FT	M	FT	MM	IN
SHALLOW-ROOTED CROPS (SPINACH, PEAS, BEANS, BEETS, CARROTS, ETC.)						
FINE SAND	100	1.2	.50	1.67	50	2.0
FINE SANDY LOAM	150	1.8	.50	1.67	75	3.0
SILT LOAM	200	2.4	.62	2.08	125	5.0
CLAY LOAM	250	3.0	.40	1.33	100	4.0
CLAY	300	3.6	.25	.83	75	3.0
MODERATELY DEEP-ROOTED CROPS (CORN, COTTON, TOBACCO, CEREAL GRAINS)						
FINE SAND	100	1.2	.75	2.50	75	3.0
FINE SANDY LOAM	150	1.8	1.00	3.33	150	6.0
SILT LOAM	200	2.4	1.00	3.33	200	8.0
CLAY LOAM	250	3.0	.80	2.67	200	8.0
CLAY	300	3.6	.50	1.67	50	6.0
DEEP-ROOTED CROPS (ALFALFA, PASTURES, SHRUBS)						
FINE SAND	100	1.2	1.00	3.33	100	4.0
FINE SANDY LOAM	150	1.8	1.00	3.33	150	6.0
SILT LOAM	200	2.4	1.25	4.17	250	10.0
CLAY LOAM	250	3.0	1.00	3.33	250	10.0
CLAY	300	3.6	.67	2.22	200	8.0
ORCHARDS						
FINE SAND	100	1.2	1.50	5.00	150	6.0
FINE SANDY LOAM	150	1.8	1.67	5.55	250	10.0
SILT LOAM	200	2.4	1.50	5.00	300	12.0
CLAY LOAM	250	3.0	1.00	3.33	250	10.0
CLAY	300	3.6	.67	2.22	200	8.0
CLOSED MATURE FOREST						
FINE SAND	100	1.2	2.50	8.33	250	10.0
FINE SANDY LOAM	150	1.8	2.00	6.66	300	12.0
SILT LOAM	200	2.4	2.00	6.66	400	16.0
CLAY LOAM	250	3.0	1.60	5.33	400	16.0
CLAY	300	3.6	1.17	3.90	350	14.0

THESE FIGURES ARE FOR MATURE VEGETATION. YOUNG CULTIVATED CROPS, SEEDLINGS, AND OTHER IMMATURE VEGETATION WILL HAVE SHALLOWER ROOT ZONES AND, HENCE, HAVE LESS WATER AVAILABLE FOR THE USE OF THE VEGETATION. AS THE PLANT DEVELOPS FROM A SEED OR A YOUNG SPROUT TO THE MATURE FORM, THE ROOT ZONE WILL INCREASE PROGRESSIVELY FROM ONLY A FEW INCHES TO THE VALUES LISTED ABOVE. USE OF A SERIES OF SOIL MOISTURE RETENTION TABLES WITH SUCCESSIVELY INCREASING VALUES OF AVAILABLE MOISTURE PERMITS THE SOIL MOISTURE TO BE DETERMINED THROUGHOUT THE GROWING SEASON.

TABLE A-6

SOIL MOISTURE RETENTION TABLE - 25 mm

Soil moisture retained after different amounts of potential evapotranspiration have occurred. Water holding capacity of soil is 25 mm.

PE	0	1	2	3	4	5	6	7	8	9
	Water Retained in Soil									
0	25	24	23	22	21	20	19	18	17	16
10	16	15	15	14	13	13	12	12	11	11
20	10	10	9	9	8	8	8	8	7	7
30	7	6	6	6	5	5	5	5	5	4
40	4	4	4	4	3	3	3	3	3	3
50	3	3	3	2	2	2	2	2	2	2
60	2	2	2	2	1	1	1	1	1	1
70	1	1	1	1	1	1	1	1	1	1
80	1	1	1	1	1	1	1	1	1	1
90	1	1	1	1						

TABLE A-7

SOIL MOISTURE RETENTION TABLE - 50 MM

Soil moisture retained after different amounts of potential evapotranspiration have occurred. Water holding capacity of soil is 50 mm.

PE	0	1	2	3	4	5	6	7	8	9
					Water Retained in Soil					
0	50	49	48	47	46	45	44	43	42	41
10	41	40	39	38	37	36	36	35	34	33
20	33	32	32	31	30	30	29	28	28	27
30	27	26	25	25	24	24	23	23	22	22
40	21	21	20	20	19	19	19	18	18	18
50	17	17	17	16	16	16	15	15	15	14
60	14	14	13	13	13	13	12	12	11	11
70	11	11	11	10	10	10	10	9	9	9
80	9	3	9	8	8	8	8	8	8	8
90	7	7	7	7	7	7	7	6	6	6
100	6	6	6	6	6	6	5	5	5	5
110	5	5	5	5	4	4	4	[illegible]	4	4
120	4	4	4	4	4	4	4	4	3	3
130	3	3	3	3	3	3	3	3	3	3
140	3	3	3	3	2	2	2	2	2	2
150	2	2	2	2	2	2	2	2	2	2
160	2	2	2	2	2	2	2	2	1	1
170	1	1	1	1	1	1	1	1	1	1
180	1	1	1	1	1	1	1	1	1	1
190	1	1	1	1	1	1	1	1	1	1
200	1	1	1	1	1	1	1	1	1	1
210	1	1	1	1	1	1	1	1	1	1
220	1	1	1	1	1					

TABLE A-8

SOIL MOISTURE RETENTION TABLE - 75 mm

Soil moisture retained after different amounts of potential evapotranspiration have occurred. Water holding capacity of soil is 75 mm.

PE	0	1	2	3	4	5	6	7	8	9
					Water Retained in Soil					
0	75	74	73	72	71	70	69	68	67	66
10	65	64	63	62	61	60	59	58	58	57
20	57	56	55	54	53	53	52	51	51	50
30	50	49	48	47	46	46	45	45	44	44
40	43	43	42	41	40	40	39	39	38	38
50	38	37	36	36	35	35	35	34	34	33
60	33	33	32	31	31	31	30	30	29	29
70	28	28	28	27	27	27	26	26	25	25
80	25	24	24	23	23	23	23	22	22	22
90	22	21	21	21	20	20	20	20	19	19
100	19	19	18	18	18	18	17	17	17	17
110	16	16	16	16	15	15	15	15	15	14
120	14	14	14	14	13	13	13	13	13	13
130	12	12	12	12	12	12	11	11	11	11
140	11	11	11	10	10	10	10	10	10	10
150	10	10	9	9	9	9	9	9	9	8
160	8	8	8	8	8	8	8	8	7	7
170	7	7	7	7	7	7	7	7	6	6
180	6	6	6	6	6	6	6	6	6	6
190	6	6	5	5	5	5	5	5	5	5
200	5	5	5	4	4	4	4	4	4	4
210	4	4	4	4	4	4	4	4	4	4
220	4	4	3	3	3	3	3	3	3	3
230	3	3	3	3	3	3	3	3	3	3
240	3	3	3	3	3	3	3	3	3	3
250	2	2	2	2	2	2	2	2	2	2
260	2	2	2	2	2	2	2	2	2	2
270	2	2	2	2	2	2	2	2	2	2
280	2	2	2	2	2	2	2	2	2	2
290	1	1	1	1	1	1	1	1	1	1
300	1	1	1	1	1	1	1	1	1	1
310	1	1	1	1	1	1	1	1	1	1
320	1	1	1	1	1	1	1	1	1	1
330	1	1	1	1	1	1	1	1	1	1
340	1	1	1	1	1	1	1	1	1	1
350	1	1	1	1	1	1	1	1	1	1
360	1	1	1	1	1	1	1	1	1	1

TABLE A-9

SOIL MOISTURE RETENTION TABLE - 100 mm

SOIL MOISTURE RETAINED AFTER DIFFERENT AMOUNTS OF POTENTIAL EVAPOTRANSPIRATION HAVE OCCURRED. WATER HOLDING CAPACITY OF SOIL IS 100 mm.

PE	0	1	2	3	4	5	6	7	8	9
					WATER RETAINED IN SOIL					
0	100	99	98	97	96	95	94	93	92	91
10	90	89	88	88	87	86	85	84	83	82
20	81	81	80	79	78	77	77	76	75	74
30	74	73	72	71	70	70	69	68	68	67
40	66	66	65	64	64	63	62	62	61	60
50	60	59	59	58	58	57	56	56	55	54
60	54	53	53	52	52	51	51	50	50	49
70	49	48	48	47	47	46	46	45	45	44
80	44	44	43	43	42	42	41	41	40	40
90	40	39	39	39	38	38	37	37	36	36
100	36	35	35	35	34	34	34	33	33	33
110	32	32	32	31	31	31	30	30	30	30
120	29	29	29	28	28	28	27	27	27	27
130	26	26	25	26	25	25	25	24	24	24
140	24	24	23	23	23	23	22	22	22	22
150	22	21	21	21	21	20	20	20	20	20
160	19	19	19	19	19	18	18	18	18	18
170	18	17	17	17	17	17	16	16	16	16
180	16	16	15	15	15	15	15	15	14	14
190	14	14	14	14	14	14	13	13	13	13
200	13	13	12	12	12	12	12	12	12	12
210	12	11	11	11	11	11	11	11	11	11
220	10	10	10	10	10	10	10	10	10	10
230	9	9	9	9	9	9	9	9	9	9
240	8	8	8	8	8	8	8	8	8	8
250	8	8	8	7	7	7	7	7	7	7
260	7	7	7	7	7	7	6	6	6	6
270	6	6	6	6	6	6	6	6	6	6
280	6	6	6	6	6	5	5	5	5	5
290	5	5	5	5	5	5	5	5	5	5
300	5	5	4	4	4	4	4	4	4	4
310	4	4	4	4	4	4	4	4	4	4
320	4	4	4	4	4	4	4	4	4	4
330	3	3	3	3	3	3	3	3	3	3
340	3	3	3	3	3	3	3	3	3	3
350	3	3	3	3	3	3	3	3	3	2
360	2	2	2	2	2	2	2	2	2	2
370	2	2	2	2	2	2	2	2	2	2
380	2	2	2	2	2	2	2	2	2	2
390	2	2	2	2	2	2	2	2	2	2
400	2	2	2	2	2	2	2	2	2	2
410	2	2	2	2	2	1	1	1	1	1
420	1	1	1	1	1	1	1	1	1	1
430	1	1	1	1	1	1	1	1	1	1
440	1	1	1	1	1	1	1	1	1	1
450	1	1	1	1	1	1	1	1	1	1
460	1	1	1	1	1	1	1	1	1	1
470	1	1	1	1	1	1	1	1	1	1
480	1	1	1	1	1	1	1	1	1	1
490	1	1	1	1	1	1	1	1	1	1
500	1	1	1	1	1	1	1	1	1	1

TABLE A-10

SOIL MOISTURE RETENTION TABLE - 125 mm

SOIL MOISTURE RETAINED AFTER DIFFERENT AMOUNTS OF POTENTIAL EVAPOTRANSPIRATION HAVE OCCURRED. WATER HOLDING CAPACITY OF SOIL IS 125 mm.

PE	0	1	2	3	4	5	6	7	8	9
				WATER RETAINED IN SOIL						
0	125	124	123	122	121	120	119	118	117	116
10	115	114	113	112	111	110	109	108	107	106
20	106	105	104	103	102	102	101	100	99	99
30	98	97	96	95	94	94	93	92	91	90
40	90	89	88	87	86	86	85	84	84	83
50	83	82	82	81	80	80	79	79	78	77
60	76	76	75	74	74	73	73	72	72	71
70	70	70	69	69	68	68	67	67	66	65
80	65	64	64	63	63	62	62	61	61	60
90	60	59	59	58	58	57	57	56	56	55
100	55	55	54	54	53	53	53	52	52	51
110	51	51	50	50	49	49	49	48	48	47
120	47	47	46	46	45	45	45	44	44	43
130	43	43	42	42	41	41	41	41	40	40
140	40	40	39	39	39	38	38	38	38	37
150	37	37	36	36	36	35	35	35	35	34
160	34	34	33	33	33	32	32	32	32	31
170	31	31	31	30	30	30	30	30	30	29
180	29	29	29	29	29	28	28	27	27	27
190	26	25	25	25	25	25	25	25	25	25
200	24	24	24	24	24	23	23	23	23	23
210	22	22	22	22	22	22	22	21	21	21
220	21	21	21	21	20	20	20	20	20	20
230	19	19	19	19	19	18	18	18	18	18
240	18	18	17	17	17	17	17	17	17	17
250	16	16	16	16	16	16	16	16	15	15
260	15	15	15	15	15	14	14	14	14	14
270	14	14	14	14	14	13	13	13	13	13
280	13	13	13	13	13	12	12	12	12	12
290	12	12	12	12	12	11	11	11	11	11
300	11	11	11	11	11	10	10	10	10	10
310	10	10	10	10	10	10	10	10	9	9
320	9	9	9	9	9	9	9	9	9	9
330	8	8	8	8	8	8	8	8	8	8
340	8	8	8	8	8	7	7	7	7	7

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	0	5		0	5		0
350	7	7	450	3	3	550	1
360	7	6	460	3	3	560	1
370	6	6	470	3	3	570	1
380	6	5	480	2	2	580	1
390	5	5	490	2	2	590	1
400	5	5	500	2	2	600	1
410	4	4	510	2	2	610	1
420	4	4	520	2	2	620	1
430	4	4	530	2	2	630	1
440	3	3	540	2	1	640	1

TABLE A-11

SOIL MOISTURE RETENTION TABLE - 150 mm

Soil moisture retained after different amounts of potential evapotranspiration have occurred. Water holding capacity of soil is 150 mm.

PE	0	1	2	3	4	5	6	7	8	9
					Water Retained in Soil					
0	150	149	148	147	146	145	144	143	142	141
10	140	139	138	137	136	135	134	133	132	131
20	131	130	129	128	127	127	126	125	124	123
30	122	122	121	120	119	118	117	116	115	114
40	114	113	113	112	111	111	110	109	108	107
50	107	106	106	105	104	103	103	102	101	100
60	100	99	98	97	97	97	96	95	94	93
70	93	92	92	91	90	90	89	89	88	87
80	87	86	86	85	84	84	84	83	83	82
90	82	81	81	80	79	79	78	77	77	76
100	76	75	75	75	74	74	73	72	72	71
110	71	71	70	70	69	69	68	68	67	67
120	66	66	66	65	65	64	64	63	63	62
130	62	62	61	61	60	60	60	59	59	58
140	58	58	57	57	56	56	55	55	54	54
150	54	53	53	53	52	52	52	52	51	51
160	51	51	50	50	50	49	49	48	48	47
170	47	47	47	46	46	46	45	45	45	44
180	44	44	44	43	43	43	42	42	42	41
190	41	41	41	40	40	40	40	39	39	39
200	39	38	38	38	37	37	37	37	36	36
210	36	36	35	35	35	35	35	34	34	34
220	34	34	33	33	33	33	33	32	32	32
230	32	31	31	31	31	31	30	30	30	30
240	30	29	29	29	29	29	28	28	28	28
250	28	27	27	27	27	27	26	26	26	26
260	26	26	25	25	25	25	25	24	24	24
270	24	24	24	23	23	23	23	23	23	23
280	22	22	22	22	22	22	22	22	21	21
290	21	21	21	20	20	20	20	20	20	20
300	20	19	19	19	19	19	19	19	18	18
310	18	18	18	18	18	18	18	17	17	17
320	17	17	17	17	17	17	17	16	16	16
330	16	16	16	16	16	16	16	15	15	15
340	15	15	15	15	15	15	14	14	14	14
350	14	14	14	14	14	14	14	13	13	13
360	13	13	13	13	13	13	13	12	12	12
370	12	12	12	12	12	12	12	12	11	11
380	11	11	11	11	11	11	11	11	11	11
390	11	11	11	10	10	10	10	10	10	10
400	10	10	10	10	10	10	10	10	9	9
410	9	9	9	9	9	9	9	9	9	9
420	9	9	9	8	8	8	8	8	8	8
430	8	8	8	8	8	8	8	8	8	8
440	8	8	8	7	7	7	7	7	7	7

TABLE A-11 (Continued)

PE	0	1	2	3	4	5	6	7	8	9
Water Retained in Soil										
450	7	7	7	7	7	7	7	7	7	7
460	7	7	7	7	6	6	6	6	6	6
470	6	6	6	6	6	6	6	6	6	6
480	6	6	6	6	6	5	6	6	5	5
490	5	5	5	5	5	5	5	5	5	5
500	5	5	5	5	5	5	5	5	5	5
510	5	5	5	5	5	5	5	5	4	4
520	4	4	4	4	4	4	4	4	4	4
530	4	4	4	4	4	4	4	4	4	4
540	4	4	4	4	4	4	4	4	4	4
550	4	4	4	4	4	4	4	3	3	3
560	3	3	3	3	3	3	3	3	3	3
570	3	3	3	3	3	3	3	3	3	3
580	3	3	3	3	3	3	3	3	3	3
590	3	3	3	3	3	3	3	3	3	3
600	3	3	3	3	3	2	2	2	2	2
610	2	2	2	2	2	2	2	2	2	2
620	2	2	2	2	2	2	2	2	2	2
630	2	2	2	2	2	2	2	2	2	2
640	2	2	2	2	2	2	2	2	2	2
650	2	2	2	2	2	2	2	2	2	2
660	2	2	2	2	2	2	2	2	2	2
670	2	2	2	2	2	2	2	2	2	2
680	2	2	1	1	1	1	1	1	1	1
690	1	1	1	1	1	1	1	1	1	1
700	1	1	1	1	1	1	1	1	1	1
710	1	1	1	1	1	1	1	1	1	1
720	1	1	1	1	1	1	1	1	1	1
730	1	1	1	1	1	1	1	1	1	1
740	1	1	1	1	1	1	1	1	1	1

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	0	5		0	5		0	5
750	1	1	790	1	1	830	1	1
760	1	1	800	1	1	840	1	1
770	1	1	810	1	1			
780	1	1	820	1	1			

TABLE A-12

SOIL MOISTURE RETENTION TABLE - 200 mm

Soil moisture retained after different amounts of potential evapotranspiration have occurred. Water holding capacity of soil is 200 mm.

PE	0	1	2	3	4	5	6	7	8	9
					Water Retained in Soil					
0	200	199	198	197	196	195	194	193	192	191
10	190	189	188	187	186	185	184	183	182	182
20	181	180	179	178	177	176	175	174	173	173
30	172	171	170	169	168	168	167	166	165	164
40	163	162	162	161	160	159	158	158	157	156
50	155	154	153	153	152	151	151	150	149	148
60	148	147	146	145	145	144	143	142	142	141
70	140	140	139	138	138	137	136	135	135	134
80	133	133	132	131	131	130	129	129	128	127
90	127	126	125	125	124	124	123	122	122	121
100	120	120	119	119	118	118	117	116	116	115
110	115	114	113	113	112	112	111	110	110	109
120	109	108	108	107	107	106	106	105	104	104
130	104	103	102	102	102	101	100	100	99	99
140	98	98	97	97	96	96	96	95	94	94
150	94	93	93	92	92	91	91	90	90	89
160	89	88	88	88	87	87	86	86	85	85
170	85	84	84	83	83	82	82	82	81	81
180	80	80	80	79	79	78	78	78	77	77
190	76	76	76	75	75	74	74	74	73	73
200	73	72	72	71	71	71	70	70	70	69
210	69	69	68	68	68	67	67	66	66	66
220	66	65	65	65	64	64	64	63	63	63
230	62	62	62	61	61	61	60	60	60	60
240	59	59	59	58	58	58	58	57	57	57
250	56	56	56	56	55	55	55	54	54	54
260	54	53	53	53	52	52	52	52	51	51
270	51	51	50	50	50	50	49	49	49	49
280	48	48	48	48	47	47	47	47	46	46
290	46	46	46	45	45	45	45	44	44	44
300	44	44	43	43	43	43	42	42	42	42
310	42	41	41	41	41	41	40	40	40	40
320	40	39	39	39	39	39	38	38	38	38
330	38	37	37	37	37	37	36	36	36	36
340	36	36	35	35	35	35	35	34	34	34
350	34	34	34	33	33	33	33	33	32	32
360	32	32	32	32	32	32	31	31	31	31
370	31	30	30	30	30	30	30	29	29	29
380	29	29	29	29	29	28	28	28	28	28
390	28	28	27	27	27	27	27	27	27	26
400	26	26	26	25	25	25	26	25	25	25
410	25	25	25	25	25	24	24	24	24	24
420	24	24	24	23	23	23	23	23	23	23
430	23	22	22	22	22	22	22	22	22	22
440	22	21	21	21	21	21	21	21	21	21

TABLE A-12 (Continued)

WATER RETAINED IN SOIL

PE	0	1	2	3	4	5	6	7	8	9
450	20	20	20	20	20	20	20	20	20	20
460	20	19	19	19	19	19	19	19	19	19
470	18	18	18	18	18	18	18	18	18	18
480	18	18	17	17	17	17	17	17	17	17
490	17	17	16	16	16	16	16	16	16	16
500	16	16	16	16	16	16	15	15	15	15
510	15	15	15	15	15	15	15	15	14	14
520	14	14	14	14	14	14	14	14	14	14
530	14	14	14	13	13	13	13	13	13	13
540	13	13	13	13	13	13	13	12	12	12
550	12	12	12	12	12	12	12	12	12	12
560	12	12	12	12	12	11	11	11	11	11
570	11	11	11	11	11	11	11	11	11	11
580	11	10	10	10	10	10	10	10	10	10
590	10	10	10	10	10	10	10	10	10	10
600	10	9	9	9	9	9	9	9	9	9
610	9	9	9	9	9	9	9	9	9	9
620	9	9	9	9	8	8	8	8	8	8
630	8	8	8	8	8	8	8	8	8	8
640	8	8	8	8	8	8	8	8	8	8
650	7	7	7	7	7	7	7	7	7	7
660	7	7	7	7	7	7	7	7	7	7
670	7	7	7	7	7	7	6	6	6	6
680	6	6	6	6	6	6	6	6	6	6
690	6	6	6	6	6	6	6	6	6	6
700	6	6	6	6	6	6	6	6	6	6
710	6	6	5	5	5	5	5	5	5	5
720	5	5	5	5	5	5	5	5	5	5
730	5	5	5	5	5	5	5	5	5	5
740	5	5	5	5	5	5	5	5	5	5
750	5	5	5	5	5	5	5	5	5	5
760	5	5	5	5	5	5	5	5	5	5
770	4	4	4	4	4	4	4	4	4	4
780	4	4	4	4	4	4	4	4	4	4
790	4	4	4	4	4	4	4	4	4	4
800	4	4	4	4	4	4	4	4	4	4
810	4	4	4	4	4	4	4	4	4	4
820	4	4	3	3	3	3	3	3	3	3
830	3	3	3	3	3	3	3	3	3	3
840	3	3	3	3	3	3	3	3	3	3
850	3	3	3	3	3	3	3	3	3	3
860	3	3	3	3	3	3	3	3	3	3
870	3	3	3	3	3	3	3	3	3	3
880	3	3	3	3	2	2	2	2	2	2
890	2	2	2	2	2	2	2	2	2	2

TABLE A-12 (Concluded)

PE	0	1	2	3	4	5	6	7	8	9
					WATER RETAINED IN SOIL					
900	2	2	2	2	2	2	2	2	2	2
910	2	2	2	2	2	2	2	2	2	2
920	2	2	2	2	2	2	2	2	2	2
930	2	2	2	2	2	2	2	2	2	2
940	2	2	2	2	2	2	2	2	2	2
950	2	2	2	2	2	2	2	2	2	2
960	2	2	2	2	2	2	2	2	2	2
970	2	2	2	2	2	2	2	2	2	2
980	2	2	2	2	2	2	2	2	2	2
990	2	2	2	2	2	1	1	1	1	1
1000	1	1	1	1	1	1	1	1	1	1
1010	1	1	1	1	1	1	1	1	1	1
1020	1	1	1	1	1	1	1	1	1	1
1030	1	1	1	1	1	1	1	1	1	1
1040	1	1	1	1	1	1	1	1	1	1
1050	1	1	1	1	1	1	1	1	1	1

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	0	5		0	5		0	5
1060	1	1	1110	1	1	1160	1	1
1070	1	1	1120	1	1	1170	1	1
1080	1	1	1130	1	1	1180	1	1
1090	1	1	1140	1	1	1190	1	1
1100	1	1	1150	1	1			

TABLE A-13

SOIL MOISTURE RETENTION TABLE - 250 MM

SOIL MOISTURE RETAINED AFTER DIFFERENT AMOUNTS OF POTENTIAL EVAPOTRANSPIRATION HAVE OCCURRED. WATER HOLDING CAPACITY OF SOIL IS 250 MM.

PE	0	1	2	3	4	5	6	7	8	9
					WATER RETAINED IN SOIL					
0	250	249	248	247	246	245	244	243	242	241
10	240	239	238	237	236	235	234	233	232	231
20	231	230	229	228	227	226	225	224	223	222
30	222	221	220	219	218	217	216	215	214	213
40	213	212	211	210	209	208	208	207	206	205
50	204	204	203	202	201	200	199	198	197	196
60	196	195	194	193	192	192	191	190	189	188
70	188	188	187	186	185	185	184	183	182	181
80	181	180	179	178	177	177	176	176	175	174
90	174	173	172	171	171	170	170	169	169	168
100	167	167	166	165	165	164	164	163	162	161
110	160	160	159	159	158	157	157	156	156	155
120	154	154	153	152	152	151	151	150	149	148
130	148	147	146	146	145	145	144	144	143	143
140	142	142	141	140	140	139	139	138	137	137
150	136	136	135	135	134	134	133	132	132	131
160	131	130	130	129	129	128	128	127	127	126
170	126	125	125	124	124	123	123	122	122	121
180	121	120	120	119	119	118	118	117	117	116
190	116	115	115	114	114	114	113	113	112	112
200	111	111	110	110	109	109	108	108	108	107
210	107	107	106	106	105	105	105	104	104	103
220	103	103	102	102	101	101	101	100	100	99
230	99	99	98	98	97	97	97	96	96	95
240	95	95	94	94	93	93	93	92	92	91
250	91	91	90	90	89	89	89	88	88	87
260	87	87	87	86	86	86	85	85	85	84
270	84	84	83	83	82	82	82	82	81	81
280	81	81	80	80	79	79	79	79	78	78
290	78	78	77	77	76	76	76	76	75	75
300	74	74	74	73	73	73	73	72	72	71
310	71	71	71	70	70	70	70	69	69	68
320	69	68	68	68	67	67	67	66	66	66
330	66	66	65	65	65	65	64	64	64	63
340	63	63	63	62		62	62	61	61	61
350	61	61	60	60	60	60	59	59	59	58
360	58	58	58	57	57	57	57	56	56	56
370	56	55	55	55	55	55	54	54	54	54
380	54	54	53	53	53	53	53	52	52	52
390	52	52	51	51	51	51	51	50	50	50
400	50	50	49	49	49	49	49	48	48	48
410	48	48	47	47	47	47	47	46	46	46
420	46	46	45	45	45	45	45	44	44	44
430	44	44	43	43	43	43	43	42	42	42
440	42	42	42	42	41	41	41	41	41	41

TABLE A-13 (Continued)

PE	0	1	2	3	4	5	6	7	8	9
WATER RETAINED IN SOIL										
450	41	41	40	40	40	40	40	40	40	40
460	39	39	39	39	39	38	38	38	38	38
470	37	37	37	37	37	37	37	36	36	36
480	36	36	36	35	35	35	35	35	35	35
490	34	34	34	34	34	34	34	34	33	33
500	33	33	33	33	33	32	32	32	32	32
510	32	32	32	32	31	31	31	31	31	31
520	31	31	30	30	30	30	30	30	30	30
530	29	29	29	29	29	29	29	29	28	28
540	28	28	28	28	28	28	28	28	27	27
550	27	27	27	27	27	26	26	26	26	26
560	26	26	26	26	26	26	26	25	25	25
570	25	25	25	25	25	24	24	24	24	24
580	24	24	24	24	24	24	24	23	23	23
590	23	23	23	23	23	23	23	23	22	22
600	22	22	22	22	22	22	22	22	21	21
610	21	21	21	21	21	21	21	21	20	20
620	20	20	20	20	20	20	20	20	20	20
630	20	20	20	19	19	19	19	19	19	19
640	19	19	19	19	19	18	18	18	18	18
650	18	18	18	18	18	18	18	18	17	17
660	17	17	17	17	17	17	17	17	17	17
670	17	17	17	17	16	16	16	16	16	16
680	16	16	16	16	16	16	16	16	15	15
690	15	15	15	15	15	15	15	15	15	15
700	15	15	15	15	15	14	14	14	14	14
710	14	14	14	14	14	14	14	14	14	14
720	14	14	14	14	13	13	13	13	13	13
730	13	13	13	13	13	13	13	13	13	13
740	13	13	13	12	12	12	12	12	12	12
750	12	12	12	12	12	12	12	12	12	12
760	12	12	12	12	11	11	11	11	11	11
770	11	11	11	11	11	11	11	11	11	11
780	11	11	11	11	10	10	10	10	10	10
790	10	10	10	10	10	10	10	10	10	10
800	10	10	10	10	10	10	10	10	9	9
810	9	9	9	9	9	9	9	9	9	9
820	9	9	9	9	9	9	9	9	9	9
830	9	9	9	9	8	8	8	8	8	8
840	8	8	8	8	8	8	8	8	8	8
850	8	8	8	8	8	8	8	8	8	8
860	8	8	8	8	8	8	8	8	7	7
870	7	7	7	7	7	7	7	7	7	7
880	7	7	7	7	7	7	7	7	7	7
890	7	7	7	7	7	7	7	7	7	7

TABLE A-13 (Concluded)

WATER RETAINED IN SOIL

PE	0	1	2	3	4	5	6	7	8	9
900	6	6	6	6	6	6	6	6	6	6
910	6	6	6	6	6	6	6	6	6	6
920	6	6	6	6	6	6	6	6	6	6
930	6	6	6	6	6	6	6	6	6	6
940	6	6	6	6	6	6	6	5	5	5
950	5	5	5	5	5	5	5	5	5	5
960	5	5	5	5	5	5	5	5	5	5
970	5	5	5	5	5	5	5	5	5	5
980	5	5	5	5	5	5	5	5	5	5
990	5	5	5	5	5	5	5	5	5	5
1000	4	4	4	4	4	4	4	4	4	4
1010	4	4	4	4	4	4	4	4	4	4
1020	4	4	4	4	4	4	4	4	4	4
1030	4	4	4	4	4	4	4	4	4	4
1040	4	4	4	4	4	4	4	4	4	4
1050	4	4	4	4	4	4	4	4	4	4
1060	3	3	3	3	3	3	3	3	3	3
1070	3	3	3	3	3	3	3	3	3	3
1080	3	3	3	3	3	3	3	3	3	3
1090	3	3	3	3	3	3	3	3	3	3
1100	3	3	3	3	3	3	3	3	3	3
1110	3	3	3	3	3	3	3	3	3	3
1120	3	3	3	3	3	3	3	3	3	3
1130	3	3	3	3	3	2	2	2	2	2
1140	2	2	2	2	2	2	2	2	2	2
1150	2	2	2	2	2	2	2	2	2	2
1160	2	2	2	2	2	2	2	2	2	2
1170	2	2	2	2	2	2	2	2	2	2
1180	2	2	2	2	2	2	2	2	2	2
1190	2	2	2	2	2	2	2	2	2	2
1200	2	2	2	2	2	2	2	2	2	2
1210	2	2	2	2	2	2	2	2	2	2
1220	2	2	2	2	2	2	2	2	2	2
1230	2	2	2	2	2	2	2	2	2	2
1240	2	2	2	2	2	2	2	2	2	2
1250	2	2	2	2	2	2	2	2	2	2

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	0	5		0	5		0	5
1260	2	2	1310	1	1	1400	1	1
1270	1	1	1320	1	1	1450	1	1
1280	1	1	1330	1	1	1500	1	1
1290	1	1	1340	1	1	1550	1	1
1300	1	1	1350	1	1			

TABLE A-14

SOIL MOISTURE RETENTION TABLE - 300 MM

SOIL MOISTURE RETAINED AFTER DIFFERENT AMOUNTS OF POTENTIAL EVAPOTRANSPIRATION HAVE OCCURRED. WATER HOLDING CAPACITY OF SOIL IS 300 MM.

PE	0	1	2	3	4	5	6	7	8	9
					WATER RETAINED IN SOIL					
0	300	299	298	297	296	295	294	293	292	291
10	290	289	288	287	286	285	284	283	282	281
20	280	279	278	278	277	276	275	274	273	272
30	271	270	269	268	268	267	266	265	264	263
40	262	261	260	260	259	258	257	256	255	254
50	254	253	252	251	250	249	248	248	247	246
60	245	244	244	243	242	241	240	240	239	238
70	237	236	236	235	234	233	232	232	231	230
80	229	228	228	227	225	225	225	224	223	222
90	222	221	220	219	219	218	217	216	215	215
100	214	214	213	212	212	211	210	209	209	208
110	207	207	206	205	204	204	203	202	202	201
120	200	200	199	198	198	197	196	196	195	194
130	194	193	192	192	191	191	190	189	189	188
140	187	187	186	186	185	184	184	183	182	182
150	181	181	180	179	179	178	178	177	176	176
160	175	175	174	173	173	172	172	171	171	170
170	170	169	168	168	167	167	166	166	155	164
180	164	163	163	162	162	161	160	160	159	159
190	158	158	157	157	156	156	155	155	154	154
200	153	153	152	152	151	151	150	150	149	149
210	148	148	147	147	146	146	145	145	144	144
220	143	143	142	142	141	141	140	140	139	139
230	138	138	138	137	137	136	136	135	135	134
240	134	133	133	132	132	132	131	131	130	130
250	130	129	128	128	128	127	127	126	125	126
260	125	125	124	124	124	123	123	122	122	121
270	121	121	120	120	119	119	119	118	118	117
280	117	117	116	116	115	115	115	114	114	114
290	113	113	112	112	112	111	111	110	110	110
300	109	109	109	108	108	108	107	107	106	106
310	106	105	105	105	104	104	104	103	103	103
320	102	102	102	101	101	101	100	100	100	99
330	99	98	98	98	98	97	97	97	96	96
340	96	95	95	95	94	94	94	93	93	93
350	92	92	92	92	91	91	91	90	90	90
360	89	89	89	88	88	88	88	87	87	87
370	86	86	86	86	85	85	85	84	84	84
380	84	83	83	83	82	82	82	82	81	81
390	81	80	80	80	80	80	79	79	79	79
400	78	78	78	77	77	77	77	76	76	76
410	76	75	75	75	74	74	74	74	74	73
420	73	73	72	72	72	72	72	71	71	71
430	71	70	70	70	70	70	69	69	69	68
440	68	68	68	68	67	67	67	67	66	66

TABLE A-14 (Continued)

PE	0	1	2	3	4	5	6	7	8	9
WATER RETAINED IN SOIL										
450	66	66	66	65	65	65	65	64	64	64
460	64	64	63	63	63	63	63	62	62	62
470	62	62	61	61	61	61	61	60	60	60
480	60	60	59	59	59	59	59	58	58	58
490	58	58	57	57	57	57	57	56	56	56
500	56	56	55	55	55	55	55	54	54	54
510	54	54	54	53	53	53	53	53	52	52
520	52	52	52	52	51	51	51	51	51	50
530	50	50	50	50	50	50	49	49	49	49
540	49	49	48	48	48	48	48	48	47	47
550	47	47	47	47	46	46	46	46	46	46
560	46	45	45	45	45	45	45	44	44	44
570	44	44	44	44	44	43	43	43	43	43
580	43	42	42	42	42	42	42	42	42	41
590	41	41	41	41	41	41	40	40	40	40
600	40	40	40	39	39	39	39	39	39	39
610	38	38	38	38	38	38	38	38	38	37
620	37	37	37	37	37	37	36	36	36	36
630	36	36	36	36	36	36	35	35	35	35
640	35	35	35	34	34	34	34	34	34	34
650	34	34	33	33	33	33	33	33	33	33
660	32	32	32	32	32	32	32	32	32	32
670	32	31	31	31	31	31	31	31	31	31
680	30	30	30	30	30	30	30	30	30	30
690	30	29	29	29	29	29	29	29	29	29
700	28	28	28	28	28	28	29	28	28	28
710	28	27	27	27	27	27	27	27	27	27
720	27	26	26	26	26	26	26	26	26	26
730	26	26	26	26	25	25	25	25	25	25
740	25	25	25	25	25	24	24	24	24	24
750	24	24	24	24	24	24	24	24	23	23
760	23	23	23	23	23	23	23	23	23	23
770	22	22	22	22	22	22	22	22	22	22
780	22	22	22	22	22	21	21	21	21	21
790	21	21	21	21	21	21	21	21	20	20
800	20	20	20	20	20	20	20	20	20	20
810	20	20	20	20	20	19	19	19	19	19
820	19	19	19	19	19	19	19	19	19	18
830	18	18	18	18	18	18	18	18	18	18
840	18	18	18	18	18	18	18	17	17	17
850	17	17	17	17	17	17	17	17	17	17
860	17	17	17	17	16	16	16	16	16	16
870	16	16	16	16	16	16	16	16	16	16
880	16	16	16	16	15	15	15	15	15	15
890	15	15	15	15	15	15	15	15	15	15

TABLE A-14 (Concluded)

PE	0	1	2	3	4	5	6	7	8	9
	Water Retained in Soil									
900	15	15	14	14	14	14	14	14	14	14
910	14	14	14	14	14	14	14	14	14	14
920	14	14	14	14	13	13	13	13	13	13
930	13	13	13	13	13	13	13	13	13	13
940	13	13	13	13	12	12	12	12	12	12
950	12	12	12	12	12	12	12	12	12	12
960	12	12	12	12	12	12	12	12	12	12
970	12	12	11	11	11	11	11	11	11	11
980	11	11	11	11	11	11	11	11	11	11
990	11	11	11	11	11	11	11	10	10	10
1000	10	10	10	10	10	10	10	10	10	10
1010	10	10	10	10	10	10	10	10	10	10
1020	10	10	10	10	10	10	10	10	10	10
1030	9	9	9	9	9	9	9	9	9	9
1040	9	9	9	9	9	9	9	9	9	9
1050	9	9	9	9	9	9	9	9	9	9
1060	8	8	8	8	8	8	8	8	8	8
1070	8	8	8	8	8	8	8	8	8	8
1080	8	8	8	8	8	8	8	8	8	8
1090	8	8	8	8	8	8	8	8	8	8
1100	7	7	7	7	7	7	7	7	7	7
1110	7	7	7	7	7	7	7	7	7	7
1120	7	7	7	7	7	7	7	7	7	7
1130	7	7	7	7	7	7	7	7	7	7
1140	6	6	6	6	6	6	6	6	6	6
1150	6	6	6	6	6	6	6	6	6	6
1160	6	6	6	6	6	6	6	6	6	6
1170	6	6	6	6	6	6	6	6	6	6
1180	6	6	6	6	6	6	6	6	6	6
1190	6	6	6	6	6	5	5	5	5	5

……

	0	5		0	5		0	5
1200	5	5	1300	4	4	1400	3	3
1210	5	5	1310	4	4	1410	3	3
1220	5	5	1320	4	4	1420	2	2
1230	5	5	1330	3	3	1430	2	2
1240	5	5	1340	3	3	1440	2	2
1250	4	4	1350	3	3	1450	2	2
1260	4	4	1360	3	3	1460	2	2
1270	4	4	1370	3	3	1470	2	2
1280	4	4	1380	3	3	1480	2	2
1290	4	4	1390	3	3	1490	2	2

TABLE A-15

SOIL MOISTURE RETENTION TABLE - 350 MM

SOIL MOISTURE RETAINED AFTER DIFFERENT AMOUNTS OF POTENTIAL EVAPOTRANSPIRATION HAVE OCCURRED. WATER HOLDING CAPACITY OF SOIL IS 350 MM.

PE	0	1	2	3	4	5	6	7	8	9
					WATER RETAINED IN SOIL					
0	350	349	348	347	346	345	344	343	342	341
10	340	339	338	337	336	335	334	333	332	331
20	330	329	328	327	326	326	324	323	322	321
30	321	320	319	318	317	316	316	315	314	313
40	312	311	310	309	308	307	307	306	305	304
50	303	302	301	300	299	299	298	297	296	295
60	294	294	293	292	291	290	289	288	287	286
70	286	285	284	283	282	282	281	280	279	278
80	278	277	276	275	274	274	273	272	271	270
90	270	269	268	257	266	256	265	264	263	263
100	262	252	261	260	259	259	258	257	256	255
110	255	254	253	252	251	251	250	250	249	248
120	248	247	246	245	244	244	243	243	242	241
130	241	240	239	239	237	237	236	236	235	234
140	234	233	233	232	231	231	230	229	228	227
150	227	226	226	225	224	224	223	223	222	221
160	221	220	220	219	218	218	217	217	216	215
170	215	214	214	213	212	211	211	210	209	209
180	208	208	207	207	206	205	205	205	204	203
190	203	202	202	201	200	200	199	199	198	197
200	197	196	196	195	194	194	193	193	192	191
210	191	190	190	189	189	188	188	187	187	188
220	186	185	185	184	183	183	182	182	181	181
230	181	180	179	179	178	178	178	177	176	176
240	175	175	174	174	173	173	173	172	171	171
250	170	170	169	169	168	168	167	167	166	166
260	166	155	165	164	164	163	163	162	162	161
270	161	161	160	160	159	159	158	158	157	517
280	156	156	155	155	154	154	154	153	153	152
290	152	152	151	151	150	150	150	149	149	148
300	148	148	147	147	146	145	145	144	144	143
310	143	143	142	142	141	141	141	140	140	139
320	139	139	138	138	137	137	137	136	136	135
330	135	135	135	134	134	134	133	133	133	132
340	132	132	131	131	130	130	130	129	129	129
350	128	128	127	127	126	126	126	125	125	124
360	124	124	123	123	122	122	122	122	121	121
370	121	121	120	120	120	119	119	119	118	118
380	117	117	117	116	116	116	116	115	115	115
390	114	114	114	113	113	112	112	112	111	111
400	111	111	110	110	110	109	109	109	108	108
410	108	108	107	107	107	106	106	106	105	105
420	104	104	104	103	103	103	103	102	102	102
430	102	102	101	101	101	100	100	100	99	99
440	99	99	98	98	98	97	97	97	96	96

TABLE A-15 (Continued)

PE	0	1	2	3	4	5	6	7	8	9
					WATER RETAINED IN SOIL					
450	96	96	95	95	95	94	94	94	94	93
460	93	93	93	92	92	92	92	91	91	91
470	90	90	90	90	89	89	89	89	88	88
480	88	88	88	87	87	87	87	86	86	86
490	85	85	85	85	84	84	84	84	84	83
500	83	83	83	83	82	82	82	82	81	81
510	81	81	81	80	80	80	80	79	79	79
520	78	78	78	78	77	77	77	77	76	76
530	76	76	76	75	75	75	75	75	74	74
540	74	74	74	73	73	73	73	73	72	72
550	72	72	72	71	71	71	71	71	70	70
560	70	70	70	69	69	69	69	69	68	68
570	68	68	68	67	67	67	67	67	66	66
580	66	66	66	65	65	65	65	65	65	64
590	64	64	64	64	63	63	53	63	63	63
600	62	62	62	62	62	61	61	61	61	61
610	60	60	60	60	60	50	60	59	59	59
620	59	59	59	58	58	58	58	58	58	57
630	57	57	57	57	56	56	56	56	56	56
640	56	56	55	55	55	55	55	55	54	54
650	54	54	54	53	53	53	53	53	52	52
660	52	52	52	52	52	52	52	51	51	51
670	51	51	51	51	50	50	50	50	50	50
680	50	50	50	49	49	49	49	49	48	48
690	48	48	48	48	47	47	47	47	47	47
700	47	47	47	46	46	46	46	46	46	46
710	45	45	45	45	45	45	45	45	44	44
720	44	44	44	43	43	43	43	43	43	43
730	43	43	43	42	42	42	42	42	42	42
740	42	42	41	41	41	41	41	41	41	41
750	40	40	40	40	40	40	40	40	40	40
760	39	39	39	39	39	39	39	39	39	38
770	38	38	38	38	38	38	38	38	38	37
780	37	37	37	37	37	37	37	37	37	36
790	36	36	36	36	36	36	36	36	35	35
800	35	35	35	35	35	34	34	34	34	34
810	34	34	34	34	34	34	34	33	33	33
820	33	33	33	33	33	33	33	33	32	32
830	32	32	32	32	32	32	32	32	32	31
840	31	31	31	31	31	31	31	31	31	30
850	30	30	30	30	30	30	30	30	30	30
860	30	30	30	29	29	29	29	29	29	29
870	29	29	29	28	28	28	28	28	28	28
880	28	28	28	28	27	27	27	27	27	27
890	27	27	27	27	27	27	27	26	25	26

TABLE A_15 (Concluded)

WATER RETAINED IN SOIL

PE	0	1	2	3	4	5	6	7	8	9
900	26	26	26	26	26	26	26	26	26	26
910	26	26	26	25	25	25	25	25	25	25
920	25	25	25	25	24	24	24	24	24	24
930	24	24	24	24	24	24	24	24	24	23
940	23	23	23	23	23	23	23	23	23	23
950	23	23	23	23	22	22	22	22	22	22
960	22	22	22	22	22	22	22	22	22	22
970	22	22	22	21	21	21	21	21	21	21
980	21	21	21	21	21	21	21	21	20	20
990	20	20	20	20	20	20	20	20	20	20
1000	20	20	20	20	19	19	19	19	19	19
1010	19	19	19	19	19	19	19	19	19	19
1020	19	19	19	18	18	18	18	18	18	18
1030	18	18	18	18	18	18	18	18	18	18
1040	18	18	18	18	17	17	17	17	17	17
1050	17	17	17	17	17	17	17	17	17	17
1060	17	17	17	16	16	16	16	16	16	16
1070	16	16	16	16	16	16	16	16	16	16
1080	16	16	16	15	15	15	15	15	15	15
1090	15	15	15	15	15	15	15	15	15	15
1100	15	15	15	15	15	15	15	15	14	14
1110	14	14	14	14	14	14	14	14	14	14
1120	14	14	14	14	14	14	14	14	14	14
1130	14	14	14	13	13	13	13	13	13	13
1140	13	13	13	13	13	13	13	13	13	13

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	0	5		0	5		0	5
1150	13	13	1300	8	8	1450	5	5
1160	12	12	1310	8	8	1500	5	5
1170	12	12	1320	8	8	1550	4	4
1180	12	12	1330	8	8	1600	4	4
1190	11	11	1340	8	8	1650	3	3
1200	11	11	1350	7	7	1700	3	3
1210	11	11	1360	7	7	1750	2	2
1220	10	10	1370	7	7	1800	2	2
1230	10	10	1380	7	7	1850	2	2
1240	10	10	1390	6	6	1900	2	2
1250	10	10	1400	6	6	1950	1	1
1260	9	9	1410	6	6	2000	1	1
1270	9	9	1420	6	6	2100	1	1
1280	9	9	1430	6	6	2200	1	1
1290	8	8	1440	6	6			

TABLE A-16

SOIL MOISTURE RETENTION TABLE - 400 mm

SOIL MOISTURE RETAINED AFTER DIFFERENT AMOUNTS OF POTENTIAL EVAPOTRANSPIRATION HAVE OCCURRED. WATER HOLDING CAPACITY OF SOIL IS 400 mm.

PE	0	1	2	3	4	5	6	7	8	9
					WATER RETAINED IN SOIL					
0	400	399	398	397	396	395	394	393	392	391
10	390	389	388	387	386	385	384	383	382	381
20	380	379	378	377	376	376	375	374	373	372
30	371	370	369	368	367	366	366	365	364	363
40	362	361	360	359	359	357	357	356	355	354
50	353	352	351	350	349	348	347	347	346	345
60	344	344	343	342	341	340	339	338	337	336
70	335	335	334	333	332	331	330	329	328	327
80	327	326	325	324	323	323	322	321	320	319
90	319	318	317	316	315	315	314	313	312	311
100	311	310	309	308	307	307	306	305	304	303
110	303	303	302	301	300	300	299	298	297	296
120	296	295	294	293	292	292	291	290	289	288
130	288	288	287	286	285	285	284	283	282	282
140	281	281	280	279	279	278	277	276	275	275
150	274	274	273	272	271	271	270	269	268	268
160	267	267	266	265	264	264	264	263	262	261
170	261	260	260	259	258	257	257	256	255	255
180	254	254	253	252	252	251	251	250	249	248
190	248	248	247	246	245	245	244	244	243	242
200	242	242	241	240	239	239	238	238	237	236
210	236	236	235	234	233	233	233	232	231	231
220	230	230	229	229	228	227	227	226	225	225
230	224	224	223	223	222	222	221	221	220	219
240	219	218	218	217	216	216	216	215	215	214
250	213	213	213	212	211	211	210	210	209	209
260	208	208	207	207	206	205	205	204	204	203
270	203	202	202	201	200	200	200	199	199	198
280	198	197	197	196	196	195	195	194	194	193
290	193	192	192	191	191	190	190	189	189	188
300	188	188	187	187	186	186	185	185	184	184
310	183	183	182	182	181	181	181	180	180	179
320	179	179	179	178	177	177	176	176	175	175
330	174	174	174	173	173	172	172	172	171	171
340	170	170	169	169	168	168	168	167	167	166
350	166	166	165	165	164	164	164	163	163	162
360	162	162	161	161	160	160	160	159	159	158
370	158	158	157	157	156	156	156	155	155	154
380	154	154	153	153	152	152	152	151	151	150
390	150	150	149	149	148	148	148	147	147	146
400	146	146	145	145	145	144	144	144	143	143
410	143	142	142	142	141	141	141	140	140	140
420	139	139	139	138	138	137	137	137	136	136
430	136	136	135	135	134	134	134	133	133	133
440	132	132	132	131	131	131	130	130	130	129

TABLE A-16 (Continued)

WATER RETAINED IN SOIL

PE	0	1	2	3	4	5	6	7	8	9
450	129	129	128	128	128	127	127	127	126	126
460	126	126	125	125	124	124	124	124	123	123
470	123	122	122	122	121	121	121	121	120	120
480	120	120	119	119	119	118	118	118	117	117
490	117	117	116	116	116	115	115	115	114	114
500	114	114	113	113	113	112	112	112	111	111
510	111	111	110	110	110	110	109	109	109	108
520	108	108	108	107	107	107	107	106	106	106
530	105	105	105	105	104	104	104	104	103	103
540	103	103	102	102	102	102	101	101	101	101
550	100	100	100	100	99	99	99	99	99	98
560	98	98	98	97	97	97	96	96	96	96
570	95	95	95	95	94	94	94	94	93	93
580	93	93	93	92	92	92	92	91	91	91
590	91	91	90	90	90	90	89	89	89	89
600	88	88	88	88	88	87	87	87	87	87
610	86	86	86	86	86	85	85	85	85	85
620	84	84	84	84	84	83	83	83	83	83
630	82	82	82	82	82	81	81	81	81	81
640	80	80	80	80	80	79	79	79	79	79
650	78	78	78	78	78	77	77	77	77	77
660	76	76	76	75	76	75	75	75	75	75
670	74	74	74	74	74	73	73	73	73	73
680	72	72	72	72	72	71	71	71	71	71
690	70	70	70	70	70	70	69	69	69	69
700	59	68	68	68	68	58	68	68	67	67
710	67	67	67	67	66	66	56	66	66	66
720	65	65	65	65	65	65	64	64	64	64
730	64	64	64	63	63	63	63	63	62	62
740	62	62	62	62	61	61	61	61	61	61
750	61	50	60	60	60	60	60	60	60	59
760	59	59	59	59	59	58	58	58	58	58
770	58	58	58	57	57	57	57	57	57	56
780	56	56	56	56	56	56	55	55	55	55
790	55	55	55	54	54	54	54	54	54	54
800	54	54	53	53	53	53	53	53	52	52
810	52	52	52	52	52	52	52	51	51	51
820	51	51	51	50	50	50	50	50	50	50
830	50	50	49	49	49	49	49	49	49	48
840	48	48	48	48	48	48	48	48	47	47
850	47	47	47	47	47	46	46	46	46	46
860	46	46	46	45	45	45	45	45	45	45
870	45	45	45	44	44	44	44	44	44	44
880	44	44	44	43	43	43	43	43	43	43
890	43	43	43	42	42	42	42	42	42	42

TABLE A-16 (Continued)

PE	0	1	2	3	4	5	6	7	8	9
					WATER RETAINED IN SOIL					
900	42	42	42	41	41	41	41	41	41	41
910	41	40	40	40	40	40	40	40	40	40
920	39	39	39	39	39	39	39	39	39	39
930	38	38	38	38	38	38	38	38	38	38
940	38	38	37	37	37	37	37	37	37	37
950	37	37	37	36	36	36	36	36	36	36
960	36	36	36	35	35	35	35	35	35	35
970	35	35	35	34	34	34	34	34	34	34
980	34	34	34	34	34	34	34	33	33	33
990	33	33	33	33	33	33	33	33	32	32
1000	32	32	32	32	32	32	32	32	32	32
1010	32	31	31	31	31	31	31	31	31	31
1020	31	31	31	30	30	30	30	30	30	30
1030	30	30	30	30	30	30	30	30	29	29
1040	29	29	29	29	29	29	29	29	29	29
1050	28	28	28	28	28	28	28	28	28	28
1060	28	28	28	27	27	27	27	27	27	27
1070	27	27	27	27	27	27	27	27	26	26
1080	26	26	26	26	26	26	26	26	26	26
1090	26	26	26	26	25	25	25	25	25	25
1100	25	25	25	25	25	25	25	25	25	25
1110	24	24	24	24	24	24	24	24	24	24
1120	24	24	24	24	24	24	24	24	24	23
1130	23	23	23	23	23	23	23	23	23	23
1140	23	23	23	22	22	22	22	22	22	22
1150	22	22	22	22	22	22	22	22	22	22
1160	22	22	22	21	21	21	21	21	21	21
1170	21	21	21	21	21	21	21	21	21	21
1180	21	21	20	20	20	20	20	20	20	20
1190	20	20	20	20	20	20	20	20	20	20
1200	20	20	19	19	19	19	19	19	19	19
1210	19	19	19	19	19	19	19	19	19	19
1220	19	19	19	18	18	18	18	18	18	18
1230	18	18	18	18	18	18	[illegible]	18	18	18
1240	18	18	18	17	17	17	17	17	17	17
1250	17	17	17	17	17	17	17	17	17	17
1260	17	17	17	17	17	17	17	16	16	16
1270	16	16	16	16	16	16	16	16	16	16
1280	16	16	16	16	16	16	16	16	15	15
1290	16	16	16	15	15	15	15	15	15	15
1300	15	15	15	15	15	15	15	15	15	15
1310	15	15	15	15	15	15	15	15	15	15
1320	14	14	14	14	14	14	14	14	14	14
1330	14	14	14	14	14	14	14	14	14	14
1340	14	14	14	14	14	14	14	14	14	14

TABLE A-16 (Concluded)

PE	0	1	2	3	4	5	6	7	8	9
					WATER RETAINED IN SOIL					
1350	13	13	13	13	13	13	13	13	13	13
1360	13	13	13	13	13	13	13	13	13	13
1370	13	13	13	13	13	13	13	13	13	13
1380	12	12	12	12	12	12	12	12	12	12
1390	12	12	12	12	12	12	12	12	12	12
1400	12	12	12	12	12	12	12	12	12	12
1410	12	12	12	12	12	11	11	11	11	11
1420	11	11	11	11	11	11	11	11	11	11
1430	11	11	11	11	11	11	11	11	11	11
1440	11	11	11	11	11	11	11	11	11	11
1450	10	10	10	10	10	10	10	10	10	10
1460	10	10	10	10	10	10	10	10	10	10
1470	10	10	10	10	10	10	10	10	10	10
1480	10	10	10	10	10	10	10	10	10	10
1490	9	9	9	9	9	9	9	9	9	9
1500	9	9	9	9	9	9	9	9	9	9
1510	9	9	9	9	9	9	9	9	9	9
1520	9	9	9	9	9	9	9	9	9	9
1530	8	8	8	8	8	8	8	8	8	8
1540	8	8	8	8	8	8	8	8	8	8
1550	8	8	8	8	8	8	8	8	8	8

……

	0	5		0	5		0	5
1560	8	8	1750	5	5	2250	1	1
1570	8	8	1800	4	4	2300	1	1
1580	8	8	1850	4	4	2350	1	1
1590	7	7	1900	3	3	2400	1	1
1600	7	7	1950	3	3	2450	1	1
1620	7	7	2000	3	3	2500	1	1
1640	6	6	2050	2	2	2550	1	1
1660	6	6	2100	2	2	2600	1	1
1680	6	6	2150	2	2	2650	1	1
1700	6	6	2200	2	2			

APPENDIX B. CASE STUDY SITES USED FOR EVALUATION OF LEACHATE GENERATION MODELS.

TABLE B-1. SUMMARY OF SELECTED CONDITIONS AT CASE STUDY SITES (SCS ENGINEERS, 1976)

Site	General Location	General Physical Features**	Surface Area		Average Depth of Fill		Daily Waste Amt.		Age of Site yrs	Waste Type rec'd %‡	Type of Fill
			ha	ac	M	ft	tonne	ton			
A	Northwest (State of Washington)	Site part of glacial outwash valley; stratigraphy composed of glacial tills, outwash gravels, and silty clays; Soil permeability = 10^{-6}; Average surface slope = 2.5%	23	58	10.0	33	907	1000	10*	R 60 (est) C 30 (est) I 10 (est)	area
B	West (State of California)	Area of metavolcanic origin with well defined bedrock. Soils consist of clayey gravels and outwash sand/gravels; Soil permeability = 10^{-3}; Average surface slope = 2%	10	25	6.1	20	91	100	20†	R 65 (est) C 25 (est) I 10 (est)	canyon
C	Midwest (State of Indiana)	Area composed of bedrock of blue shale/limestone; overlain by silty clays; Soil permeability = 10^{-5}; Average surface slope = 7.6%	0.4	1	6.1	20	15	17	3*	R 50 C 50 I 0	trench area
D	East (State of Pennsylvania)	Area of igneous intrusion with diabasic and limestone bedrock; silty loam soils; Soil permeability = 10^{-6}; Average surface slope = 10%	4	11	2.5	8	87	96	2*,#	R 48 C 21 I 31	area
E	South (State of Tennessee)	Area of fractured Cambrian shale bedrocks overlain by weathered shales and clays; Soil permeability = 10^{-6}; Average surface slope = 10%	3	7	6.1	20	86	95	2†	R 88 C 9 I 3	canyon and trench

* Operated as a landfill from its inception.

† Originally operated as a burning dump; now operated as a landfill.

# Filling activities complete; site now closed.

** Permeability values shown are for soil used as cover material, cm/sec.

‡ R = Residential, C = Commercial, I = Industrial.

TABLE B-2. MEASURED DATA OF LEACHATE GENERATION, PRECIPITATION, AND EVAPORATION.* (SCS Engineers, 1976)

Month / Site	A			B			C			D			E		
	L**	P‡	E##	L	P	E	L**	P‡	E##	L	P	E	L**	P‡	E##
January	---†	---	---	0.13	3.42	1.05	0.87	1.19	0	---†	---	---	---†	---	---
February	0.06	7.34	---	0.16#	12.22	0.70	0.10	4.84	0	0.01	0.78	0	0.08	4.35	1.17
March	0.10	5.79	2.02	0.20#	14.20	1.99	0.10	5.44	0	0.02#	2.68	0	0.19	9.12	3.07
April	0.18	4.82	2.46	0.23	2.43	3.70	0.10	3.22	2.93	0.03	2.52	1.90	0.08	2.30	4.29
May	0.18	2.45	4.34	0.07	0.28	8.36	2.33	2.01	6.63	0.04	2.45	4.82	0.07	4.63	4.47
June	0.03	2.41	4.44	0.14	0.22	9.20	1.17	6.25	6.73	0.04	5.55	5.60	0.13	3.53	6.00
July	0.03	0.75	5.20	0.08	0.67	10.38	0.68	3.51	7.43	0.03	6.47	6.48	0.04	2.90	6.56
August	0.07	5.43	3.70	0.01	0.45	9.16	0.39	4.02	5.62	0.06	3.71	5.37	0.01	2.35	5.81
September	0.03	0	3.02	0.02	0	7.67	0.49	3.75	2.92	0.01	8.01	3.46	0.13	3.33	3.80
October	0.27	---	---	0.22	5.13	2.80	0.58	6.24	3.51	0.03	3.05	2.30	0.01	3.29	2.38
Total (inches)	0.95	28.99	25.24	1.26	39.02	55.01	6.81	40.47	35.77	0.27	35.22	29.93	0.74	35.88	37.55
L,ave, per mo.	0.11	---	---	0.13	---	---	.68	---	---	0.03	---	---	0.082	---	---

* All units in inches

† No data available

# Estimated

** Measured Leachate Flow, inches/mo.

‡ Measured precipatation, inches/mo.

## Measured precipatation, inches/mo.

APPENDIX C. MUNICIPAL LANDFILL LEACHATE COMPOSITION

TABLE C-1. MUNICIPAL LANDFILL LEACHATE COMPOSITION
(Values in mg/l except as noted)

Researcher	Hughes et al. (1971) Blackwell	Hughes et al. (1971) LW 5B Dupage	Hughes et al. (1971) LW 6B Dupage	Pohland (1975) control cell
Age of Refuse	initial	6 years	17 years	1 year
Type of Study	landfill	landfill	landfill	FTC
Constituents				
COD	39,680 (sic)	8,000 (sic)	40 (sic)	9,000
BOD$_5$	54,610 (sic)	14,080 (sic)	225 (sic)	5,300
TOC		6.3		2,430
pH (pH unit)			7.0	5.33
E_h (millivolts)				5,602
TS				
TDS	19,144	5,794	1,198	
TSS				1,243
VSS				602
TVS				
FS				
Sp. Cond. (μmhos/cm)				
Alk. as $CaCO_3$	3,255	4,159	1,011	1,500
Hard. as $CaCO_3$	7,830	2,200	540	1,576
Total P				
o-P				
PO_4-inor.	6	1.20	8.90	0.5
NH_4-N				80
NO_3+NO_2-N	1.70 nitrate	0.70 nitrate	1.50	6.4 nitrate
Org.-N				124
TKN				
Ca				350
Mg				39
Na	900	810	74	130
K				
SO_4	680	2	2	
Cl	1,697	1,330	135	143
Fe (total)	5.500	6.3	0.6	292
Mn "	1.66	0.06	0.06	10
Zn "		0.13	<0.10	
Cu "	0.05	<0.05	<0.05	
Cd "	<0.05	<0.05	<0.05	
Pb "		0.50	0.50	
Hg "				
Se "	2.7	<0.10	<0.10	
Cr "	0.20 (hexa)	0.15 (hexa)	<0.50 (hexa)	
Ni "				
Turb. (Jackson units)				

TABLE C-1. MUNICIPAL LANDFILL LEACHATE COMPOSITION
(Values in mg/l except as noted)

Researcher	Pohland (1975) control cell	Pohland (1975) control cell	Ministry (1961)	Merz (1954)
Age of Refuse	2 years	3 years	n.i.	1.5 years
Type of Study	FTC	FTC	landfill	FTC
Constituents				
COD	3,490	4,175		
BOD_5	2,090	1,937	5,491	31-33,100
TOC	1,098	1,320		
pH (pH unit)	5.8	5.8	7.4	5.6-7.6
E_h (millivolts)				
TS				
TDS				
TSS	340	36		
VSS	200	25		
TVS				
FS				
Sp. Cond. (µmhos/cm)				
Alk. as $CaCO_3$	1,360	760		730-9,500
Hard. as $CaCO_3$	1,192	1,342		550-3,120
Total P				0.2-29
o-P				
PO_4 -inor.	1.2	2.0		
NH_4-N	13	2.0	668	0.2-890
NO_3+NO_2-N	0.04	0.16		
Org.-N	25	14.0	101	
TKN				
Ca	150	112	248	115-2,570
Mg	22	20		64-410
Na	50	15		85-1,305
K				28-1,860
SO_4				
Cl	200	30	1845	96-2,350
Fe (total)	400	438		6.5-336
Mn "	4.5	5.40		
Zn "	13	18.7		
Cu "				
Cd "				
Pb "				
Hg "				
Se "				
Cr "				
Ni "				
Turb. (Jackson units)				

TABLE C-1. MUNICIPAL LANDFILL LEACHATE COMPOSITION
(Values in mg/l except as noted)

Researcher	Emcon (1974) control	Emcon (1974) recirculate	Chain & DeWalle (1977) University of Illinois	Reinhart and Ham (1971) MUNC
Age of Refuse	2 years	2 years	0.25 years	0.33 years
Type of Study	FTC	FTC	Landfill	Laboratory
Constituents				
COD	3,250-20,300	22,700-89,520	49,300	71,680
BOD_5	3,250-19.200	15,500-33,600	24,700	57,00
TOC			17,060	27,700
pH (pH unit)	4.7-5.4	4.6-6.5	5.63	5.97
E_h (millivolts)			-60	-132
TS			33,989	55,348
TDS	724-14,080	14,500-21,010		
TSS	16-200	22-600	139	202
VSS				
TVS				
FS			15,586	22,395
Sp. Cond. (µmhos/cm)			13,700	16,800
Alk. as $CaCO_3$	240-3,920	1,580-7,300	568	459
Hard. as $CaCO_3$				
Total P	0-2.8	1.4-79.2	21.5	98
o-P			6.5	29
PO_4 -inor.				
NH_4-N	0-81	194-880	392.5	1028
NO_3+NO_2-N	0-0.90	0-6.34	0.5	10.29
Org.-N			545	945
TKN				
Ca	20-1,082	561-1,800	3,750	3,300
Mg	2.6-608	316-672	650	1,140
Na	80-338	408-1,010	1,360	1,580
K	48-148	260-910	1,140	2,300
SO_4	20-250	257-1,040	1,110	1,558
Cl	56-490	920-1,565	1,480	2,467
Fe (total)	22-1,050	165-300	2,200	1,046
Mn "				
Zn "	0.23-9.0	6.5-35	104	370
Cu "	0.2-0.44	<0.2-0.4	0.48	0.75
Cd "	0-<.05	<0.05-0.16	0.1	0.375
Pb "	<0.5-1.81	<0.0-2.0	1.25	1.65
Hg "	0-0.013	0-0.064		
Se "				
Cr "			18	0.5
Ni "			13	2.6
Turb. (Jackson units)			75	30

TABLE C-1. MUNICIPAL LANDFILL LEACHATE COMPOSITION
(Values in mg/l except as noted)

Researcher	Fungaroli (1971)	Qasim and Burchinal (1970)	Zenone *et al.* (1974) Borough	Chun *et al.* (1975)
Age of Refuse	2 years	0.33 years	15 years	Initial
Type of Study	Laboratory	FTC	Landfill	Laboratory
Constituents				
COD	1,000-51,000			2250
BOD_5		14,760-33,360		
TOC			860-2480	
pH (pH unit)	3.7-8.5	5.88-6.48	5.7-6.9	5.0-6.9
E_h (millivolts)				
TS	0-42,000	21,140-59,200		2760
TDS				2530
TSS	10-26,500			127
VSS				95
TVS		11,385-32,250		1270
FS				
Sp. Cond. (µmhos/cm)			2460-5350	960
Alk. as $CaCO_3$	0-9,700	10,630-20,350		1280
Hard. as $CaCO_3$	0-5,500	7,600-13,100		1410
Total P			0.04-0.18	5.3
o-P				
PO_4 -inor.	0-130	127-154		
NH_4-N	0-482	473-1106	2.1-5.9	
NO_3+NO_2-N			0.2-0.3	
Org.-N		288-1416	0-1.4	
TKN	0-177		8.4-9.2	16.5
Ca		2790-4080	160-440	291-496
Mg		176-420	64-100	40-67
Na	0-7,700	584-1439	76-140	38
K		1050-3770	4.5-6.6	94
SO_4	25-450	615-1002		
Cl	4.7-2,340	951-2310	110-170	82
Fe (total)	0-1,716	175-860	34-76	31-39
Mn "			4.8-18.0	3.5-8.4
Zn "	0-167		0.08-0.260	0.7-3.1
Cu "	0-9.9		0.010	
Cd "			0	
Pb "				
Hg "				
Se "			0-0.016	
Cr "			0.01-0.02	
Ni "				
Turb. (Jackson units)				

TABLE C-1. MUNICIPAL LANDFILL LEACHATE COMPOSITION
(Values in mg/l except as noted)

Researcher	Reinhart and Ham (1973) UMC	Johansen & Carlson (1976) Yggeseth	Johansen & Carlson (1976) Taranrod	Wigh (1979) Cell 2D
Age of Refuse	0.33 years	3.5 years	2.5 years	Initial
Type of Study	Laboratory	Landfill	Landfill	FTC
Constituents				
COD	16,380	9,425	3,455	4,594
BOD_5	5,906	5,250 (total)	2,300	
TOC	9,960	1,700	800	
pH (pH unit)	5.59	5.9	6.2	5.4
E_h (millivolts)	-220			
TS	7,930	4,160	3,160	
TDS				
TSS	192	466	1,079	
VSS		182	602	
TVS		2,180	1,670	
FS	3,475			
Sp. Cond. (μmhos/cm)	5,420	3,380	2,370	3,500
Alk. as $CaCO_3$	-284	39	21.5	430
Hard. as $CaCO_3$				1,350
Total P	85	7.7	1.6	11.4
o-P	85			4.4
PO_4 -inor.				
NH_4-N	347.4	227	84	73
NO_3+NO_2-N	4.29			.73
Org.-N	78.5	23	71	81
TKN		250	156	
Ca	572	400	218	223
Mg	220	54	40	42
Na	330	206	197	297
K	90	187	214	685
SO_4	77	100	100	30
Cl	474	370	340	50
Fe (total)	91	234	69	
Mn "				15
Zn "		0.65	2.65	
Cu "		0.022	0.021	
Cd "		0.0009	0.0008	
Pb "		0.01	0.015	
Hg "				
Se "				
Cr "		0.06	0.17	
Ni "		0.03	0.12	
Turb. (Jackson units)	270			

TABLE C-1. MUNICIPAL LANDFILL LEACHATE COMPOSITION
(Values in mg/l except as noted)

Researcher	Wigh (1979) Cell 20	Wigh (1979) Cell 20	Fungaroli and Steiner (1979)	Fungaroli and Steiner (1979)
Age of Refuse	3 years	5 years	2 years	4.5 years
Type of Study	FTC	FTC	Laboratory	Laboratory
Constituents				
COD	26,064	15,484	24,520	761
BOD_5	15,640			
TOC				
pH (pH unit)	5.4	5.5	5.53	5.36
E_h (millivolts)				
TS				
TDS				
TSS			595	28.3
VSS				
TVS				
FS				
Sp. Cond. (μmhos/cm)	15,500	7,168		
Alk. as $CaCO_3$	4,790	3,330		
Hard. as $CaCO_3$	3,480	2,484		
Total P				
o-P	36	14.6		
PO_4-inor.			3.8	0
NH_4-N	538	212	185	1.3
NO_3+NO_2-N	13			
Org.-N			158	8.6
TKN	697	244		
Ca	1,440	468		
Mg	160			
Na	600		704	71
K				
SO_4	558	96	299	0
Cl	777		366	35
Fe (total)	775	1,080	806	100
Mn "	50			
Zn "	26	2.1	17.1	0.63
Cu "	0.90		1.0	0
Cd "	N.D.	0.0135		
Pb "	N.D.	0.076		
Hg "				
Se "				
Cr "				
Ni "			0.18	
Turb. (Jackson units)				

TABLE C-1. MUNICIPAL LANDFILL LEACHATE COMPOSITION
(Values in mg/l except as noted)

Researcher	Fuller (1978)	W.E.S. (1978) Landfill A	W.E.S. (1978) Landfill B	W.E.S. (1978) Landfill C
Age of Refuse	6 months	15 years	4 years	3 years
Type of Study	Laboratory	Landfill	Landfill	Landfill
Constituents				
COD	150-500			
BOD_5				
TOC		21-50	22-148	16-208
pH (pH unit)	6.3-6.8			
E_h (millivolts)				
TS				
TDS				
TSS				
VSS				
TVS				
FS				
Sp. Cond. (μmhos/cm)	2,400-2,800			
Alk. as $CaCO_3$				
Hard. as $CaCO_3$				
Total P	3.9-7.9			
o-P				
PO_4 -inor.				
NH_3-N	70-190			
NO_3+NO_2-N		.05-.47	0.03-0.15	0.03-0.05
Org.-N				
TKN				
Ca	90-275	34-127	170-210	10-37
Mg		17-48.5	6.8-130	4.4-56
Na		20-72	33-75	6.7-220
K	150-950			
SO_4		23-47	44-240	8-24
Cl	93-3,900	5	<5-12	5-415
Fe (total)	48-120	3.5-18.6	0.2-20	0.08-18.20
Mn "	0.6-1.8	3-5	1-11.3	0.19-12.30
Zn "	0.1-3.4	0.24-1.36	0.03-0.7	0.06-2.73
Cu "	N.D.	<0.02	<0.02	<0.02
Cd "	N.D.	<0.03	<0.03	<0.03
Pb "	N.D.	<0.1	<0.1	0.14-0.37
Hg "	N.D.	<0.0002	<0.0003	0.0012-0.0018
Se "		<0.002	0.004-0.007	<0.02
Cr "	N.D.	<0.03	0.04	<0.03
Ni "	N.D.	0.04-0.13	0.53-0.63	0.07-1.61
Turb. (Jackson units)				

TABLE C-1. MUNICIPAL LANDFILL LEACHATE COMPOSITION
(Values in mg/l except as noted)

Researcher	Rovers and Farquhar (1973)	SCS Engineers (1976) Site A	SCS Engineers (1976) Site B	SCS Engineers (1976) Site C
Age of Refuse	1 year	Active	Active	Active
Type of Study	FTC	Landfill	Landfill	Landfill
Constituents				
COD	0-44,000	2,251-30,933	79-420	9,690-25,634
BOD_5		1,410-17,400	6-48	3,533-18,150
TOC		920-12,800	40-63	6,100-7,300
pH (pH unit)	5.3-11.5	5.37-6.11	6.34-7.10	5.54-6.7
E_h (millivolts)				
TS				
TDS	0-17,000	1,723-34,924	1,352-1,631	7,040-11,596
TSS		110-910	19-65	
VSS				
TVS				
FS				
Sp. Cond. (µmhos/cm)		1,350-13,500	2,050-2,335	5,750-9,600
Alk. as $CaCO_3$	2,000-8,000			
Hard. as $CaCO_3$		689-3,790	816-1,250	2,350-4,300
Total P				
o-P				
PO_4-inor.		0.2-15	0.05-11	0.47-16
NH_4-N	0-650	1.7-693	1.1-693	1.2-36
NO_3+NO_2-N		0.03-12	0.07-11	0.031-11
Org.-N	0-550			
TKN		61-752	5-31	34-57
Ca	0-2,800	162-1,705	105-251	1,605-2,113
Mg	0-600	37-324	68-158	282-372
Na	0-900	50-652	69-110	340-700
K				
SO_4		90-620	80-165	1-140
Cl	0-1,300	60-390	1-200	8-647
Fe (total)	0-300	74-1,126	0.8-120	111-1,300
Mn "	0-25			
Zn "		1.2-33	0.7-4.8	0.88-47
Cu "		0.01-1.13	0.01-0.38	0.01-2.75
Cd "		0.001-0.012	0.001-0.01	0.001-0.36
Pb "		0.06-0.32	0.04-0.16	0.19-0.60
Hg "		0.001-0.008	0.001-0.32	0.001-0.009
Se "		0.005-0.10	0.001-0.11	0.008-0.3
Cr "		0.025-2.04	0.001-0.19	0.01-0.46
Ni "		0.2-0.7	0.06-0.27	0.22-1.09
Turb. (Jackson units)				

TABLE C-1. MUNICIPAL LANDFILL LEACHATE COMPOSITION
(Values in mg/l except as noted)

Researcher	SCS Engineers (1976) Site D	SCS Engineers (1976) Site E	Emrich and Landon (1969)	Emrich and Landon (1969)
Age of Refuse	2 years	Active	initial	3 years
Type of Study	Landfill	Landfill	Landfill	Landfill
Constituents				
COD	7,650-17,654	845-8,569	750,000	
BOD_5	3,360-15,900	104-2,000	720,000	
TOC	5,040-5,500	740-760		
pH (pH unit)	5.3-6.65	5.75-8.42		
E_h (millivolts)				
TS				
TDS	4,120-9,507	3,443-7,508		11,254
TSS	60-642	44-479		
VSS				
TVS				
FS				
Sp. Cond. (μmhos/cm)	3,730-8,300	2,350-10,080		
Alk. as $CaCO_3$				
Hard. as $CaCO_3$	1,400-2,920	480-3,023		8,700
Total P				
o-P				
PO_4-inor.	0.3-25	0.41-11		
NH_4-N	0.30-148	380-739		
NO_3+NO_2-N	0.047-12	0.24-27.24		
Org.-N				
TKN	94-210	16-936		
Ca	220-620	98-120		
Mg	171-224	96-104		
Na	162-662	750-1,130		
K				
SO_4	10-85			940
Cl	66-475	4-874	2,000	1,000
Fe (total)	18-1,550	4.4-35	1,000	
Mn "				
Zn "	0.53-28.2	0.32-27		
Cu "	0.01-0.65	0.016-0.48		
Cd "	0.001-0.07	0.004-0.01		
Pb "	0.17-0.31	0.02-0.33		
Hg "	0.001-0.06	0.0003-0.16		
Se "	0.01-0.33	0.002-0.21		
Cr "	0.09-0.29	0.06-0.13		
Ni "	0.07-0.56	0.08-0.21		
Turb. (Jackson units)				

TABLE C-1. MUNICIPAL LANDFILL LEACHATE COMPOSITION
(Values in mg/l except as noted)

Researcher	Emrich and Landon (1969)	Apgar and Langmuir (1971) 1962 cell *	Apgar and Langmuir (1971) 1967 cell	Anderson and Dornbush (1967)
Age of Refuse	15 years	<1 year	5.3 years	n.i.
Type of Study	Landfill	FTC	FTC	Landfill
Constituents				
COD			2,062	
BOD_5		1,360	1,920	
TOC			940	
pH (pH unit)		6.5	6.46	7.1
E_h (millivolts)			-73	
TS			2,091	
TDS	2,075			
TSS			163	
VSS				
TVS				
FS			1,294	
Sp. Cond. (μmhos/cm)			1,390	
Alk. as $CaCO_3$		502	149	471
Hard. as $CaCO_3$	300			578
Total P			0.70	
o-P			0.55	
PO_4 - inor.				
NH_4-N		18	60	
NO_3+NO_2-N			0.90	
Org.-N			4.4	
TKN				
Ca		186	1,040	293
Mg		37	75	
Na		39	350	219
K		51	160	2.8
SO_4	24	50	19.5	
Cl	220	225	657	320
Fe (total)		578	218	4.3
Mn "				
Zn "				
Cu "				
Cd "				
Pb "				
Hg "				
Se "				
Cr "				
Ni "				
Turb. (Jackson units)			43	

TABLE C-1. MUNICIPAL LANDFILL LEACHATE COMPOSITION
(Values in mg/l except as noted)

Researcher	Riccio and Hyde (1971)	Meichtry (1971) 3-18-68	Meichtry (1971) 3-24-71	County of Los Angeles (1969)
Age of Refuse	n.i.	>3 years	>6 years	n. i.
Type of Study	Landfill	Landfill	Landfill	FTC
Constituents				
COD		76,800	3,042	7,130
BOD_5		10,300	308	7,050
TOC				
pH (pH unit)		5.75	7.4	5.9
E_h (millivolts)				
TS				
TDS		44,900	13,409	9,190
TSS		172	220	
VSS				
TVS				
FS				
Sp. Cond. (μmhos/cm)				
Alk. as $CaCO_3$		9,860	8,677	1,710
Hard. as $CaCO_3$	880	22,800	8,930	3,260
Total P		0.24	0.65	
o-P				
PO_4 - inor				
NH_4-N		0	270	141
NO_3+NO_2-N	8.0			5
Org.-N				
TKN				152
Ca	140	7,200	216	905
Mg	130	15,600	8,714	254
Na	650	767	1,160	350
K	520	58	440	653
SO_4	5.6	1,190	19	1,200
Cl	710	660	2,355	
Fe (total)	0	2,320	4.75	33.3
Mn "				
Zn "				
Cu "				
Cd "				
Pb "				
Hg "				
Se "				
Cr "				
Ni "				
Turb. (Jackson units)				

TABLE C-1. MUNICIPAL LANDFILL LEACHATE COMPOSITION
(Values in mg/l except as noted)

Researcher	SCS Engineers (1978) Site 1	SCS Engineers (1978) Site 2	SCS Engineers (1978) Site 5	SCS Engineers (1978) Site 6
Age of Refuse	7 years	20 years	50 years	20 years
Type of Study	Landfill	Landfill	Landfill	Landfill
Constituents				
COD	7789	723	1198	4792
BOD_5				
TOC	3440	315	420	3000
pH (pH unit)	5.8	8.2		
E_h (millivolts)				
TS				
TDS				
TSS				
VSS				
TVS				
FS				
Sp. Cond. (μmhos/cm)				
Alk. as $CaCO_3$				
Hard. as $CaCO_3$				
Total P				
o-P				
PO_4- inor.				
NH_4-N	111	11.1	3	62
NO_3+NO_2-N	< 0.4	< 0.4	<0.1	1
Org.-N				
TKN	132	124	12	78
Ca	1375	6	96	103
Mg				
Na				
K				
SO_4	220	170	183	120
Cl	270	72	47	96
Fe (total)	0.86	4.20	30	2
Mn "				
Zn "				
Cu "	< 0.20	< 0.20	0.30	< 0.20
Cd "	< 0.03	< 0.03	< 0.01	< 0.01
Pb "	< 0.20	0.4	0.2	< 0.2
Hg "	0.005	0.005	0.002	0.140
Se "				
Cr "	< 0.30	< 0.30	1.80	< 0.20
Ni "				
Turb. (Jackson units)				

TABLE C-1. MUNICIPAL LANDFILL LEACHATE COMPOSITION
(Values in mg/l except as noted)

Researcher	County of Los Angeles	Zenone *et al.* (1974) Elmendorf	SCS Engineers (1978) Site 7	Summary Range of Values
Age of Refuse	n.i.	13 years	25-30 years	
Type of Study	Landfill	Landfill	Landfill	
Constituents				
COD			3631	0-750,000
BOD_5				6-57,000
TOC		0	1300	6.3-27,700
pH (pH unit)	8.3	7.1 -7.6		3.7-11.5
E_h (millivolts)				-132-5,602
TS				0-59,200
TDS	2,000			0-44,900
TSS				10-1,243
VSS				25-602
TVS				1,270-32,250
FS				
Sp. Cond. (umhos/cm)				960-16,800
Alk. as $CaCO_3$	1,290			0-20,350
Hard. as $CaCO_3$	537			0-22,800
Total P		0.0-0.1		0-98
o-P		0		0-85
PO_4-inor.				0-154
NH_4-N		0	160	0-1,106
NO_3+NO_2-N	18	0.29-0.32	< 1	0-27.2
Org.-N		0.08-0.09		0-1,416
TKN			175	0-936
Ca	72	17-20	118	0-7,200
Mg	87	2.6-3.2		0-15,600
Na	600	3		0-7,700
K		0.3-0.4		2.8-3,770
SO_4	99	18-19	55	0-1,558
Cl	300	1.0-1.3	406(120)	0-3,900
Fe (total)	219	0-50	76	0-5,500
Mn "				0-50
Zn "		10		0-370
Cu "			5.00	0-9.9
Cd "		0	0.04	0-0.375
Pb "			2.08	0-2.0
Hg "			0.002	0-0.16
Se "		0-14		0-2.7
Cr "		0	1.19	0.01-18
Ni "				0.04-13.0
Turb. (Jackson units)				

APPENDIX D

ADDITIONAL RESEARCH NEEDS

An examination of existing literature addressing production and management of leachate from municipal landfills has revealed a number of important deficiencies. In order to identify these areas, this section will (1) list those areas requiring additional research, (2) suggest specific objectives for research in these areas, and (3) recommend generalized approaches to meet these research needs.

Suggested research approaches will be designated by one or more of the following categories:

- Basic research - identification of the processes and mechanisms occurring in a sanitary landfill and their role in leachate generation and transport.
- Bench scale - experiments performed within the laboratory, utilizing apparatus and materials which simulate landfill conditions.
- Field studies - studies at actual landfill sites which can be used in conjunction with bench scale experiments to verify results or improve their viability.

In addition to the suggested topics and methods of approach, approximate costs and time appropriations considered necessary for completion of potential projects are estimated.

A comprehensive evaluation of the literature concerning leachate generation and assessment revealed a number of areas where additional research is warranted. For purposes of clarity, these research needs are presented in terms of the following five subject areas:

- Leachate generation
- Leachate composition
- Leachate migration

- Available control technology, and
- Environmental monitoring.

LEACHATE GENERATION

An evaluation of the literature reveals that the water balance model was used as the major tool for leachate generation estimation. Over a hundred techniques are available for leachate generation estimation by using the water balance principle. Thus far, few techniques have been verified under field conditions. The applicability, accuracy, and sensitivity of leachate generation techniques are, therefore, greatly lacking. The following research needs are suggested for the techniques deemed promising. No priority is established for such research needs. In our opinion, all such techniques should be compared by field experiments. The hydrologic simulation model HSSWDS suggested by EPA, which selects SCS curve number for surface runoff, and the Penman equation for evapotranspiration estimation, should also be verified under the landfill conditions, since the data used were mainly from agricultural lands, not landfills.

1. Determine the amount of water contributed by refuse degradation

Objective--

The water contributed by refuse degradation, although not significant, can affect the refuse water absorbing capability and hasten the generation of leachate. A thorough literature search only revealed one set of data related to refuse biological water. Additional research is suggested for gathering information for different refuse and environmental conditions.

Recommended Approach--

Bench scale--Construct laboratory lysimeters packed with municipal refuse. Variables to be considered include refuse composition, refuse age, initial refuse moisture content, degradation temperature, and other environmental and chemical factors such as pH, nutrients, toxins, etc. Experiments should be conducted at selected time intervals over a period of one to two years. Data gathered can be used to estimate ΔS_R as discussed in the generalized water balance model (Table 2, Section 3).

2. Determine the permeability of refuse

Objective--

A review of the literature found that most of the refuse permeability experiments were performed under bench testing at refuse densities of less than 700 pounds per cubic yard. In field conditions, the refuse density is usually in the range of 800 to 1200 pounds per cubic yard. Since the refuse permeability affects the rate of water percolation in refuse, such information is essential and should be obtained at conditions similar to field situations. Due to the difficulty in conducting field testing for refuse permeability, only bench scale testing is suggested.

Recommended Approach--

Bench scale--Construct laboratory lysimeters which are filled with refuse having different particle sizes (shredding), and density (compaction and baling) characteristics. Measure permeability values and develop nomographs for specific refuse physical parameters.

3. Determine the surface runoff characteristics under different landfill surface conditions.

Objective--

A review of the literature indicates that the Rational Method and Curve Number Method approaches may be the best techniques for calculating surface runoff. However, the existing information for these two methods (e.g., runoff coefficients, antecedent moisture contents, hydrologic soil groups, runoff curve numbers) are mainly derived from areas other than landfills. In order to make accurate estimations, information from actual landfills should be obtained. Due to the wide varieties of landfill surface conditions (e.g., precipitation, soil type, surface slope, vegetation type), both pilot and field testing are suggested. Pilot testing can cover all the variables which are not easily obtained in the field testing (e.g., a wide range of selected controlled precipitation). Field testing can be used to verify the pilot data for larger landfill areas.

Recommended Approach--

Pilot scale--Test cells can be constructed in the laboratory or in the landfill. Variables to be considered should cover all the factors discussed in the main text for each specific technique (e.g., Rational or Curve Number Methods). Tables or nomographs should be prepared to cover all the possible landfill conditions.

Field scale--Construct test plots in selected landfills to study surface runoff. Methods, as described in most hydrology literature, can also be used for setting up the field test site. The testing method as discussed under "Surface Measurement", (Section 3 of this report) can be followed. Artificial rainfall frequency, duration, and quantity can be used to facilitate the tests. It is suggested that experimental factors be presented in a matrix or nomograph for interpretation of test results. Experimental variables should cover most of the possible landfill conditions. These data should be compared to the pilot testing results for verification or refinement of the experimental approach. Data deemed reliable can be incorporated into existing models (e.g., HSSWDS).

4. Determine evapotranspiration rates under different landfill conditions.

Objective--

A review of the literature indicates that both the Thornthwaite and Penman approaches may be used satisfactorily for landfill conditions. However, data obtained by both approaches were not generated from landfills. Some important variables (refer to Section 3) may also be omitted in both approaches. Since the evapotranspiration estimation is one of the most important steps for the water balance calculation, data generated from landfills for evapotranspiration are considered most necessary for accurate leachate generation estimations.

Recommended Approach--

Pilot scale--Pilot scale testing can be conducted by following the set-ups used in hydrological or agricultural fields. For ease in controlling the experimental variables (e.g., soil type and moisture, vegetation type, temperature, humidity, radiation, etc. as discussed in Section 3 of this report), lysimeter testing may be desired. Statistical analysis should be used for the data evaluation.

Field scale--Field scale testing for evapotranspiration is very difficult. However, in connection with other testing (e.g., surface runoff, infiltration, and soil moisture changes versus time), the evapotranspiration can be estimated. Such data can then be used for correlation with other evapotranspiration factors (e.g., temperature, soil type, radiation, etc.) by statistical means. The data should also be compared to the pilot testing results for verification or modification of the experimental approach. Tables and nomographs should be prepared to represent most landfill and environmental conditions. The effects of additional variables on evapotranspiration obtained in this research may be used to derive a modified technique (from either Thornthwaite, Penman, or other methods).

LEACHATE COMPOSITION

While considerable information has been gathered regarding landfill behavior and leaching patterns, efforts to model the quality of leachate have revealed some information deficiencies. Though the models reviewed in Section 4 demonstrate promising predictive capabilities, their application to field-scale situations requires a broadened empirical base. Aside from the empirical support needed for modeling efforts, some basic information deficiencies are evident regrading the viability of microorganisms in landfills and leachate, as well as factors governing the leaching of heavy metals from solid waste. These research needs are summarized below.

1. Determine the perisistence and viability of enteric bacteria viruses, and parasites in municipal landfills and municipal landfill leachates.

Objective--

Recent evidence has indicated the presence of bacterial species in 9 year old landfill leachates, suggesting that even older landfills pose a potential public health threat. Though several comprehensive investigations on the presence of viruses in leachate has shown viruses are rarely present, the data base should be expanded. The presence of parasites have yet to be studied. The objective of this research is to identify the public health threat from enteric bacteria, viruses, and parasites in solid waste and sewage sludge/solid waste codisposal leachates from older landfills.

Recommended Approach--

Field scale--Perform a field survey of 20-30 municipal solid waste landfills throughout the U.S., collecting refuse core samples and leachates for microbiological examination. The landfills should range from 10 to 30 years in age. Some of the landfills should represent sewage sludge/refuse codisposal operations. The microbiological examination should include major indicator groups, such as total and fecal coliform and fecal streptococci, as well as select viruses (e.g., poliovirus) and parasites (e.g., *Ascaris*).

2. Determine the importance of ionic strength, chemical activity, complexation, adsorption, and precipitation on the removal of heavy metals from landfills by leaching.

Objective--

The concentration histories of heavy metals in leachate from several studies indicate that the leaching behavior of heavy metals does not follow the pattern established by other refuse constituents. The mobility of heavy metals is known to be influenced by various physical/chemical phenomenon which are mentioned above. The objective of this research is to identify, and if possible, quantify the effects of each of these factors on the mobility of heavy metals in landfills.

Recommended Approach--

Bench scale--Construct a series (4 or 5) of laboratory lysimeter columns. One column should contain only municipal refuse, while the others contain refuse plus varying amounts of sludge containing iron, zinc, chromium, copper, cadmium, nickel, and lead from electroplating industries. Subject each column to equal moisture regimes. Collect and analyze leachates as a function of cumulative leachate volume. Identify metal species and organic and inorganic ligands. From the data, explain the presence or absence of metals in leachate in terms of equilibrium concepts modified by consideration of ionic strength and complexation.

3. Determine the potential for leachate organic compounds to complex with heavy metals and the extent to which metal concentrations can be increased above normal solubilities by complexation.

Objective--

Determine the stability formation constants of organometallic complexes based on the organic compounds likely to be present in leachate.

Recommended Approach--

Bench scale--Collect leachates from several active municipal landfills of various ages, and identify types and levels of various organic species in the leachates. Fractionate the organic phase into their respective molecular weight fractions using ultrafiltration techniques and gel permeation chromatography, as described by Chian, *et al*. (1976). Determine the heavy metal distribution in each molecular weight fraction. Develop stability formation constants of the organometallic complexes based on thermodynamic equilibria and the empirical data gathered in this and previous studies.

4. Determine the limits of industrial waste heavy metal loadings that can be sustained during codisposal without deleterious effects on leachates on the biochemical processes of stabilization. Also, identify those landfill conditions or externally added mediators which are capable of minimizing the deleterios effects on microbial populations from added metals.

Objective--

Identify the industrial sludge loadings at which the quality of municipal landfill leachates is impaired, and whether combinations of sludges and refuse can alleviate these effects.

Recommended Approach--

Bench scale--Construct a series of laboratory lysimeters packed with shredded and unshredded refuse and seed with heavy metal-laden industrial sludge in various increments. Determine the sludge/refuse ratios resulting in increased rates of metal appearance in leachate. Sludges should include common industrial metals, such as cadmium, copper, nickel, lead, chromium, and zinc. Monitoring of lysimeter temperature, gas, and leachate is essential to evaluate the effects of added sludge on stabilization processes. Lysimeter testing should also include combinations of sludges for possible synergistic effects.

5. Determine the hydraulic properties of municipal solid waste to assist model development efforts.

Objective--

Investigate the hydraulic behavior of specific components of solid waste to identify the influence of solid waste composition on moisture movement. In this way, the hydraulic properties of specific landfills can be more accurately characterized.

Recommended Approach--

Bench scale--Construct a series of laboratory lysimeters and fill with varying proportions of moisture holding materials (e.g., paper products, rags) and impermeable material (e.g., glass, metal products). Identify the hydraulic relationships among these materials and the macro-void space associated with them. Measurements of moisture content and moisture flow within the refuse would supply information that may directly support formulations of refuse hydraulic properties. Also observe the determinants of field capacity in the lysimeter.

6. Investigate the precise nature of the solubilization of contaminants in solid waste for model development.

Objective--

Develop a more broad empirical base for the generation of specific contaminants, (e.g., major organic indicators such as BOD, COD, and TOC, and inorganic contamiants such as nutrients and heavy metals) from solid waste. By garnering more empirical data, description of specific contaminant generation through models is better supported.

Recommended Approach--

Bench scale--Fill a series of tall (20 ft) laboratory columns with municipal refuse of varying density and composition. Place intermediate cover soils between successive lifts. Apply different moisture regimes to replicate columns. Collect and analyze leachates for major organic indicators, nutrients and heavy metals over a period of 4 to 5 years. A series of curves plotting concentration and mass removal as a function of cumulative leachate volumes and time should be developed.

7. Investigate the occurrence and behavior of refuse and leachate microbial populations to better understand their impact on leachates and landfill stabilization.

Objective--
A major process model described in Section 4 assumed microbes in solution, and described their action using the Monod Model. Experimental work is needed to verify or justify these assumptions.

Recommended Approach--

Bench scale--Fill a series of laboratory flasks with landfill leachates (or synthetic leachate) of various ages. Spike each with volatile fatty acids (e.g., acetic acid, propionic acid) and a bacterial seed from a landfill slime layer (at bottom of landfill). Determine degradation rates of the acids and cell concentrations at various intervals. Determine whether Monod Kinetic Model adequately describes substrate utilization in the landfill environment. If not, suggest other models of microbial activity for use in process modeling efforts.

LEACHATE MIGRATION

A review of the existing literature indicates that additional research is required for the development and evaluation of leachate migration models which can be used to predict the extent and rate of biodegradation. Additional studies are also required to determine the effects of flow rate (or solution flux) and soil chemical and physical properties on the predictive capabilities of leachate migration models. Such efforts may lead to the development of a simplified model which can be utilized by landfill designers and site operators in estimating the attenuation of leachate constituents. These research needs are summarized below.

1. Develop a leachate migration model which incorporates a biodegradation rate constant which can be used to predict the extent and rate of biological decomposition.

Objective--
Numerous models are available for predicting chemical and physical migration; however, the integration of biodegradation models with mass transport approaches are greatly lacking. Additional research is required for the development of biodegradation models which would be of significant value in predicting the migration of biological contaminants.

Recommended Approach--

Basic research--Select several potential biodegradation models (e.g., Monod, Teissier, Contois, Moser, and various competitive/non-competitive inhibition forms) and evaluate them in terms of various fluid velocity and mass transport approaches. Combine the most promising combinations of mass transport and biodegradation models which would provide the most definitive predictive capability.

Bench scale--Construct a laboratory test column containing a soil having a low organic content. Operate the column anaerobically under continuous saturated flow conditions. Aerobiosis may be achieved by sealing the system and flushing with nitrogen after each feed addition. Apply a continuous flow of leachate of known composition to the column and measure selected biological parameters at various time intervals. Determine by statistical methods the predictive capability of different models combinations for various leachate strengths.

2. Determine the predictive capability of single-ion and multi-ion simulation models for different flow rates and constituent concentrations.

Objective--

Leachate migration models, in general, are developed by selecting the flow rate (or solution flux) as an independent variable because available data indicate that attenuation is sensitive to solution flux. The application of such models for different flow conditions is largely unknown. Additional research is required to determine if parameters estimated at one flux are comparible to those at another flux. Model validation over a range of flow rates is considered essential for practical applicability.

Recommended Approach--

Basic research--Review the literature and identify those single-ion models which may be applicable for integration with a mass transport equation. Models which may be used for a group of single or multi-ions should also be identified and evaluated for compatibility with mass transport equations.

Bench test--Construct laboratory lysimeters containing soils of varying permeabilities (e.g., clay, silty loam, sandy loam, sand). Operate the columns under continuous saturated flow conditions. Apply leachates of known composition and measure the effluent concentrations of selected water quality parameters (based on ion specific model) at various time intervals. Determine the predictive capability (at selected confidence levels) for different flow rates and solution concentrations. Based on the results of this study, modify (refine) the model(s) for imporved predictive capability.

3. Develop a simplified model based on soil key parametric indicators. Assess the effects of such simplification upon the predictive capability of the developed model.

Objective--

Most leachate migration models require complex approaches which are both unwieldly and impractical. Therefore, additional research is recommended for the development of simplified models based upon soil parametric indicators. Such development is essential for increased utilization of predictive migration models.

Recommended Approach--

Basic research--Review the literature for data correlations between soil chemical and physical properties, and model development. Construct a matrix or nomographs for data interpretation and selection of soil properties to be evaluated in bench scale testing.

Bench scale--Construct laboratory lysimeters containing soils having a range of chemical and physical properties which are known to affect the accuracy of the model. Key parameters may include texture (clay content), content of hydrous oxides, type and content of organic matter, cation exchange capacity, and permeability. Operate the columns under continuous saturated flow conditions. Apply solutions of known composition and determine the extent and rate of attenuation for selected constituents. Quantify adsorption and decay constants, dispersion coefficients, etc. Correlate these results with applicable soil characteristics. Develop a simplified model based on key controlling parameters. Verify the model over a range of flow rates and solution concentrations for different soils. Evaluate, stastically, the effects of model simplification on the accuracy of the obtained results.

LEACHATE CONTROL

A review of the literature indicates that additional studies are required to develop the technology and determine the effectiveness of various leachate control measures, particularly in the areas of overland flow leachate recycling, leachate extraction and well injection, nutrient and alkaline industrial waste addition, natural and synthetic sorbent materials, and leachate recirculation techniques. Additional research in these areas will provide for alternative leachate control approaches which can be utilized by landfill designers and operators. Research needs are summarized below.

1. Investigate and evaluate the efficiency of overland flow leachate recirculating systems in reducing the organic loading of landfill leachate.

Objective--

Operational overland flow leachate recirculating systems are not reported in the literature; however, overland flow irrigation is an inexpensive well-developed technique. Research is suggested for the development of such a system which may represent a viable means of treating recycled leachate.

Recommended Approach--

Field study--Construct an overland flow spreading basin at a sanitary landfill facility which is equipped with recycling capabilities. Evaluate the time-dependent removal efficiency of biological indicators (e.g., COD, TOC). Relate the results to the rate and frequency of recyle, and soil cover material. Estimate the rate of landfill stabilization. Assess feasible design configuration and operating procedures together with associated costs for implementation.

2. Investigate the removal efficiency and effectiveness of a leachate extraction and deep well injection system.

Objective--

Very little information is available in the literature concerning leachate extraction and deep well injection. Additional research is required to perform detailed hydrogeologic investigations to ensure that leachate is contained and does not migrate to cause contamination problems.

Recommended Approach--

Field study--Install a recycling and injection well system at an existing landfill which is producing large quantities of concentrated leachate. Determine the zone of influence for various pumping rates. Install a cluster of monitoring wells (the number dependent on the pumping sphere of influence) on the parameter of the injection well in a number of concentric rings. Monitor the migration and removal efficiency of trace metals and other applicable water quality criteria.

3. Determine the effect of injection of major nutrients (i.e., nitrogen, phosphorous) on the biological stabilization of leachate constituents.

Objective--

Available information in the literature indicates that phosphorus may be limiting in the stabilization of landfill refuse. Additional research is required to determine what nutrient ratios are required to restore or increase biological stabilization of leachate constituents.

Recommended Approach--

Bench scale--Construct a lysimeter test cell containing refuse from a landfill exhibiting a slow rate of stabilization. Develop and maintain anaerobic conditions within the cell.

Inject varying amounts of nutrients (e.g., nitrogen, phosphorous) and measure the BOD_5: N:P ratios at selected time intervals to indicate a possible limiting nutrient concentration. Determine what nutrient additions are required to restore or increase refuse stabilization as measured by the rate of reduction of COD or TOC levels.

4. Determine the effects of codisposal of municipal refuse and alkaline industrial wastes on the biodegradation of organics in landfill leachate.

Objective--

Because of the high moisture content and alkalinity of many industrial sludges, the inclusion of alklaline wastes may prove beneficial in accelerating refuse stabilization. Additional research is required to define the applicability and provide quantitative inforamtion to optimize this process.

Recommended Approach--

Bench scale--Construct lysimeter test cells and fill with municipal refuse. Add varying amounts of alkaline industrial wastes for pH adjustment. Determine the alkaline waste/refuse ratios resulting in increased refuse stabilization rates as indicated by reduction of effluent COD and TOC concentration levels. Estimate the effectiveness, application range, costs, and practicability of this method.

5. Determine the effectiveness of various natural and synthetic materials in removing contaminants from leachates.

Objective--

Laboratory studies have demonstrated the effectiveness of various sorbent materials in removing contaminants from leachates. The results indicate that combinations of different sorbents can be used to increase the attenuation of leachate contaminants. Placement of sorbents in landfills as a bottom layer to provide substantial contaminant reduction prior to migration from the site may exhibit economical feasibility due to the reduced contaminant load imposed upon the leachate treatment facilities. Additional research is required to define the applicability and optimize this approach.

Recommended Approach--

Bench scale--Construct laboratory lysimeters which are filled with a layer of various natural and synthetic materials. Selected materials should include bottom ash, fly ash, vermiculate, illite, ottowa sand, activated carbon, kaolinite, natural zeolities, activated alumina, and cullite. Apply leachates of known composition and operate the column under continuous saturated flow conditions. Measure the effluent concentration levels of trace metals and organic indicators at various time intervals and determine the removal efficiency of specific sorbent materials. Based on the results of this preliminary study, select sorbent combinations which may be more effective than individual sorbents in reducing the concentration levels of contaminants. Verify the removal efficiency of selected combinations of sorbent materials by additional bench-scale testing.

6. Investigate the effects of municipal and industrial sludge additions to municipal refuse on leachate concentration histories during leachate recirculation.

Objective--

Laboratory and field test cell investigations have demonstrated the benefits of leachate collection and recirculation back to the landfill: the rapid decline in leachate organic concentrations with time of leaching, a delay in the starting time for leachate treatment, and a leachate volume reduction through evapotranspiration. The objective of this research is to determine the adverse or beneficial effects of municipal and industrial sludge addition to refuse landfills subject to leachate recirculation.

Recommended Approach--

Bench scale--Construct a series of 5-6 laboratory columns, and provide a leachate recirculation system for each. One column would be filled with municipal refuse only, and the others refuse plus varying amounts of municipal and metal-laden industrial sludges. Subject each to identical moisture regimens, and collect periodic samples of recycled leachate for analysis. Analyze for general organic indicators (e.g., COD, BOD, TOC volatile acids) as well as heavy metals (e.g., Cu, Cd, Fe, Cr, Pb, Ni, Hg, Zn). Evaluate the effects of sludge addition on leachate recirculation and refuse stabilization.

ENVIRONMENTAL MONITORING

An evaluation of the existing literature has revealed some important deficiencies with regard to the performance of passive vadose zone monitoring devices, obtaining representative samples for analysis by use of pressure/vacuum lysimeters, and the potential environmental impacts of surface run-off from sanitary landfills. Additional studies are required in these areas to provide for the implementation of a more effective monitoring network. Research needs are summarized below.

1. Investigate and evaluate passive, non-sampling vadose zone monitoring devices underlying a sanitary landfill.

Objective--

Passive, non-sampling vadose zone monitoring devices have received limited attention which reflects, to a large extent, the complexity of flow in the vadose zone compared to saturated flow. Also, these devices experience hysteresis effects in reponse to wet-dry cycles and require frequent recalibaration depending on exposure conditions. Additional research is required for the development of improved technology for monitoring in the vadose zone.

Recommended Approach--

Bench scale--Construct laboratory lysimeters which are filled with a layer of various soils (e.g., clay, silty loam, sanding loam, fine sand), overlaid by municipal refuse, to simulate the refuse/soil interface at the bottom of a landfill. Vary the time sequence of water addition to simulate various wet/dry cycles. Calibrate the vadose zone monitoring devices against a range of soil-water contents (i.e., soil-moisture curves). Assess the sensitivity and accuracy of available passive, non-sampling devices (e.g., tensiometers, psychrometers, moisture blocks), with regard to their response to varying volumetric flow conditions.

Bench scale--Construct a laboratory lysimeter and fill with a soil having a moderate permeability (i.e., silty loam, sandy loam). Emplace moisture blocks having different encapsulation media (e.g., gypsum, fiberglass, polymeric resins). Flush the lysimeter with high ionic strength leachates to simulate the mass rate of application encountered in a 5 to 10 year period. Evaluate (by visual observation and weight-loss measurements) the effect of the applied leachate upon the lead wires and encapsulation materials. Assess the simulated time effect of the applied leachate on the accuracy and sensitivity of the respective devices by comparison of calibration curves obtained at selected time intervals.

2. Determine the validity of pressure/vacuum lysimeters for obtaining representative in situ vadose zone leachate samples. Identify water quality parameters which may be attenuated during passage through the porous cup.

Objective--

A significant problem in vadose zone monitoring is obtaining representative leachate samples. Limited laboratory studies indicate that passage of leachate through the porous cup of a pressure/vacuum lysimeter will result in attenuation of certain water quality constituents. Additional research is required to determine the extent of such attenuation for various porous cup materials.

Recommended Approach--

Bench scale--Construct an experimental test cell and emplace pressure/vacuum lysimeters constructed of different porous cup materials (e.g., ceramic, PVC, teflon). Pass a representative number of simulated or collected leachate samples through the selected porous cups. Identify, (by statistical methods), applicable water quality parameters attenuated by the cup material. Relate the results to the feasibility of obtaining representative in situ leachate samples.

3. Investigate the environmental impact of surface run-off from various municipal landfill operations.

Objective--

Surface runoff from landfill sites may result in contamination of surface water supplies. Quantitative information is generally lacking in the literature. Additional research is required to determine the effects of type of cover soil, landfill surface conditions, vegetation, and precipitation on the generation and composition of surface runoff.

Recommended Approach--

Field study--Install surface runoff collection devices at selected landfills. Select several key monitoring parameters, e.g., organic indicators, heavy metals, pathogens, and nutrients, and study their concentration and flow rate of surface runoff waters from area fill and trench disposal operations. Perform routine and synoptic storm related monitoring. Relate the results to the type of refuse, landfill surface conditions (e.g., slope, type of cover soil, vegetation), and intensity and duration of input water.

SUMMARY

The additional research needs for leachate production and associated areas are summarized in Table D-1. The specific research objectives as listed include estimates of the period of performance necessary for project completion as well as corresponding costs. The subjective cost and performance time estimates are based upon anticipated man-hour requirements and capital investment for the recommended approaches to the specific objectives as presented in the previous text.

TABLE D-1 ESTIMATED TIME AND COST EXPENDITURES FOR COMPLETION OF POTENTIAL PROJECTS

Research Area	Approach	Period of Performance	Estimated Cost ($1,000)
Leachate Generation			
Determine the amount of water contributed by refuse degradation.	Bench Scale Construct laboratory lysimeter containing refuse from representative landfills. Quantify the amount of water generation with refuse composition, refuse age, initial moisture content, temperature, pH, nutrients, toxins, etc.	1 - 2 years	100-150
Determine permeability of municipal refuse.	Bench Scale Construct laboratory lysimeters which are filled with refuse having different particle sizes (shredding), and density (compaction and baling) characteristics. Measure permeability values and develop nomographs for specific refuse physical parameters.	1 year	80
Determine surface runoff characteristics for different landfill surface conditions.	Pilot Scale Construct laboratory or landfill test cells. Measure applicable parameters for calculation of surface runoff by use of Rational or Curve Number Methods.	1 year	100
	Field Study Construct test plots at selected landfills. Follow testing methods discussed under "Surface Measurement" (Section 3 of this report). Apply simulated rainfall of varying frequency, duration, and intensity. Construct a matrix or nomographs for interpretation of test results.	1 to 2 years	100-150
Determine evapotranspiration rates under different landfill conditions.	Bench Scale Construct laboratory lysimeters for evaluation of experimental variables (e.g., soil type and moisture content, vegetation, temperature, humidity, radiation). Utilize statistical methods for data evaluation.	1 - 2 years	100-150

(Continued)

TABLE D-1 (Continued)

Research Area	Approach	Period of Performance	Estimated Cost ($1,000)
	Field Scale Construct test plots at selected landfills. Estimate evapotranspiration rates by use of water balance equations. Correlate data with other evapotranspiration factors (e.g., temperature, soil type, vegetation, radiation) by statistical methods. Compare data with bench scale studies and construct a matrix or nomographs for interpretation and application of test results	2 years	150
Leachate Composition	Field Scale		
Determine the persistence and viability of enteric bacteria, viruses, and parasites in landfills and landfill leachates.	Survey 20-30 municipal solid waste landfills, throughout the U.S., collecting refuse core samples and leachates for microbiological examination. Examine samples for bacterial indicator groups, viruses, and parasites.	1 year	125
Determine the importance of ionic strength, complexation, adsorption and precipitation on the removal of heavy metals from landfills by leaching.	Bench Scale Construct a series of laboratory columns containing varying amounts of municipal refuse and electroplating sludge. Explain the concentrations of heavy metals in the leachates in terms of equilibrium concepts modified by consideration of ionic strength and complexation.	2 years	70 - 80
Determine the stability formation constants of organometallic complexes based on the organic compounds found in leachates.	Bench Scale Collect leachates from several active municipal landfills of various age and fractionate the organic phase into respective molecular weight fractions. Develop stability formation constants of the complexes based on thermodynamic equilibria and empirical data.	6 - 8 months	50 - 60
Determine the industrial sludge loadings at which the quality of landfill leachates is impaired, and whether combinations of sludges can alleviate these effects.	Bench Scale Construct a series of laboratory lysimeters and fill with refuse and heavy-metal laden sludges in various increments. Determine sludge/refuse ratios resulting in increased levels of metals in leachates, and decreased rates of stabilization.	2 years	70 - 80

(Continued)

TABLE D-1 (Continued)

Research Area	Approach	Period of Performance	Estimated Cost ($1,000)
Determine the hydraulic behavior of specific solid waste components to identify the influence of solid waste composition on moisture movement for model development.	Bench Scale Construct a series of laboratory columns and fill with varying proportions of impermeable and moisture holding refuse materials. Measure moisture content and moisture flow, and estimate field capacities.	6 months	40 - 50
Investigate the solubilization of contaminants in solid waste to broaden the empirical base for model development.	Bench Scale Fill a series of tall laboratory columns with municipal refuse of varying density and composition. Apply different moisture regimes. Collect and analyze leachates, and develop concentration history curves.	4 - 5 years	100
Investigate the behavior of refuse and leachate microbial populations to determine their impacts on leachates and landfill stabilization for model development.	Bench Scale Using laboratory flasks filled with leachates and spiked with a bacterial seed, determine the degradation rates of volatile acids and determine cell concentrations at various intervals. Determine whether the Monod Kinetic Model describes substrate utilization in the landfill environment.	6 months	20 - 30
Leachate Migration			
Develop a leachate migration model which incorporates a biodegradation factor.	Basic Research Assess various biodegradation models and evaluate in terms of applicability when intergrated with mass transport models.	1 year	80
	Bench Scale Construct test columns containing low grade soils and determine predicitive capability in terms of the biodegradation indicator for different flow rates and leachate concentrations.	1 - 2 years	100 - 150

(Continued)

TABLE D-1 (Continued)

Research Area	Approach	Period of Performance	Estimated Cost ($1,000)
Develop ion specific models which may be used in leachate migration prediction studies.	Basic Research Review literature and identify ion specific models which are applicable in a mass transport equation.	6 months	50
	Bench Scale Construct lysimeters containing soils of various permeabilities. Apply leachates of known composition at saturated flow conditions. Evaluate the accuracy of the model for different flow rates and solution concentrations.	1 - 2 years	100 - 150
Develop simplified models based on soil parametric indicators.	Basic Research Review the literature for correlations between soil parameters indicators and model development.	1 year	80
	Bench Scale Construct laboratory lysimeters containing soils of different chemical and physical properties. Apply leachates of known composition. Refine the model based on the test results and determine the effect of simplification on predictive capability.	1 - 2 years	100 - 150
Leachate Control			
Examine removal efficiency of leachate contaminants by overland flow.	Field Study Construct overland flow plot at a sanitary landfill equipped with recycling capabilities. Evaluate removal efficiency of biological indicators. Relate the results to the rate and frequency of recycle, and soil cover for treatment.	1 - 2 years	100 - 150
Investigate the removal efficiency and effectiveness of a leachate withdrawal and injection well system.	Field Study Install a recycling system at an existing landfill. Monitor the migration and removal efficiency of key indicator parameters throughout the entire system. Install a monitoring well network on the perimeter of the injection well. Determine the zone of influence for various pumping rates.	1 - 2 years	150 - 250

(Continued)

TABLE D-1 (Continued)

Research Area	Approach	Period of Performance	Estimated Cost ($1,000)
Determine the effects of injection of major nutrients on biological stabilization and limiting nutrient concentrations.	Bench Scale Construct laboratory lysimeters containing refuse. Maintain anaerobic conditions within the test cell. Determine the effectiveness of nutrient additions on refuse stabilization as measured by the reduction in effluent COD or TOC levels.	1 - 2 years	80 - 150
Determine the effects of codisposal of municipal refuse and alkaline industrial wastes on biological stabilization.	Bench Scale Construct laboratory lysimeters containing refuse and various amounts of alkaline industrial wastes for pH adjustment. Determine alkaline waste/refuse ratios resulting in increased refuse stabilization.	1 year	80
Determine the effectiveness of selected sorbent materials in attenuating leachate contaminants.	Bench Scale Construct laboratory lysimeters containing different sorbent materials. Apply leachates of known composition. Determine the removal efficiency of trace metals and organic indicators for individual sorbents. Select sorbent combinations for improved attenuation of contaminants.	1 - 2 years	100 - 150
Determine the adverse or beneficial effects of municipal and industrial sludge. Additions to refuse landfills subject to leachate recirculation.	Bench Scale Construct a series of laboratory columns equipped with a leachate recirculation system. Fill columns with varying amounts of municipal and industrial sludge using one column as a control. Collect recycled leachate periodically for analysis. Evaluate effects of sludge addition on leachate recirculation.	2 years	100

(Continued)

TABLE D-1 (Continued)

Research Area	Approach	Period of Performance	Estimated Cost ($1,000)
Environmental Monitoring			
Investigate and evaluate passive, non-sampling vadose zone, monitoring devices.	Bench Scale Construct lysimeters and place vadose monitoring devices below simulated refuse/soil interface. Assess sensitivity and accuracy for various wet/dry cycles.	1 year	80
	Bench Scale Emplace moisture blocks of different encapsulation materials within lysimeters filled with neutral pH soil. Flush with high ionic strength leachate and assess the simulated long-term equipment performance.	6 months	50
Determine the validity of pressure/vacuum lysimeters for obtaining representative vadose zone leachate samples.	Bench Scale Select commercially available lysimeters constructed of various porous cup materials. Pass representative leachate solution through the porous cups to determine the attenuation of water quality parameters.	1 year	50
Investigate the impact of surface runoff from active landfill sites.	Field Study Install surface runoff collection devices at selected landfills. Perform routine and synoptic storm related monitoring. Relate the results to the type of cover soil, landfill surface conditions, vegetation, intensity, and duration of input water.	1 - 2 years	80 - 120